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Chemistry of Carcass Burial Leachate

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UNIVERSITY OF SASKATCHEWAN

Final Report

This final report covers results of the livestock mortality burial study at the University of Saskatchewan over the period of August 2005-December 2007. The study consists of:

- Five lined burial pits filled with livestock mortalities
- One pit contains poultry, two contain swine and two contain bovine.
- One of the swine pits consists of swine layered with soil with a soil cover, while the other swine pit contains only swine and is covered with polyethylene with soil on top, bovine is configured the same way
- Chemical analysis of leachate gathered during the two year study
- Contaminant transport models using MIGRATEv9 and Geo-Slope CTRAN 2004

The report is provided as an excerpt of Dyan Pratt's M.Sc. thesis titled *Environmental Impact of Livestock Mortalities Burial.* At the time of this report, Ms. Pratt had not defended the thesis and therefore, although this a final report and should be considered at such, the thesis is considered in draft form and some minor edits are expected. Ms. Pratt is expected to defend the thesis in March 2008 and a final draft of the thesis will be available by April 2008. Additionally, two journal articles are being prepared from the thesis material. One paper regarding the leachate chemistry and one regarding the groundwater transport modeling.

Publications and presentations related to this study

Publications:

Pratt, D.L & T.A. Fonstad. 2006. Livestock Mortalities Burial Leachate Chemistry: The First Nine Months. ASABE Intersectional Meeting. Saskatoon, SK. Paper # MBSK 06-303.

(Pending)

D.L. Pratt, 2008. Environmental Impact of Livestock Mortalities Burial. M.Sc. Thesis, Department of Agricultural and Bioresources Engineering, University of Saskatchewan. May 2008(pending). 113pp.

Presentations:

Saskatchewan Agriculture and Food. Agricultural Operations research update. December 2006.

Pratt, Dyan. 2006. Livestock Mortalities Burial Leachate Chemistry: The first nine months. Presented at the ASAE/CSAE Annual Intersectional Meeting, October, 2006. Saskatoon, SK, Canada.

ENVIRONMENTAL IMPACT OF LIVESTOCK MORTALITIES BURIAL

An Excerpt of a Thesis Submitted to the College of
Graduate Studies and Research
In Partial Fulfillment of the Requirements
For the Degree of Master of Science
In the Department of Agricultural & Bioresource Engineering
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By

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Keywords: livestock burial

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ABSTRACT

The objective of this thesis was to determine the potential groundwater impact as a result of leachate from livestock mortality burial for three species of livestock: swine, bovine and poultry. Specific objectives were to:

1. Determine the chemical composition of leachate in livestock mortality burial pits for three species: bovine, swine and poultry
2. Develop simulations for contaminant transport using two different contaminant transport models MIGRATEv9 and Geo-Slope CTRAN
3. Based on the simulations, provide worst case scenario transport distances

Livestock mortality leachate contains on average concentrations of 9000 mg/l of ammonium-N, 31000 mg/l alkalinity (as bicarbonate), 2500 mg/l chloride, 4500 mg/l sulphate, 2000 mg/l potassium, 1900 mg/l sodium, 1700 mg/l phosphorus along with relative smaller amounts of iron, calcium and magnesium. The pH of the leachate was near neutral. In comparison to earthen manure storages and landfills, the strength of the leachate was 2-4 times higher.

Preliminary simulations were created with two software packages, MIGRATEv9 and Geo-Slope CTRAN to simulate transport of the leachate for three different soil conditions using a conservative tracer. Transport through a clayey soil was dominated by diffusion allowing the leachate to transport a distance of approximately three meters in 50 years. The till soil situation produced a transport depth of six meters with an approximate concentration of thirty percent of initial concentration in 50 years. For a sandy soil, transport reached a depth of 10 meters in 10 years with approximately forty

percent of initial concentration. The MIGRATE and Geo-Slope models produced similar results with the MIGRATE model predicting slightly higher concentration values than that of Geo-Slope.

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1 INTRODUCTION

Canada's agricultural industry plays an important role in livestock production throughout the world. Canada is home to approximately 16.2 million head of cattle, 14.5 million swine, 1.7 million sheep, 625 million chickens and 20.5 million turkeys (Statistics Canada 2007). Of the 16.2 million cattle, over half reside in Saskatchewan and Alberta. Livestock populations can easily be threatened by foreign disease outbreaks, natural disasters and agro terrorism. Diseases such as avian flu, hoof and mouth, and transmissible spongiform encephalopathy (TSE) also known as bovine spongiform encephalopathy (BSE) or mad cow in cattle, chronic wasting disease in deer and elk and Creutzfeldt-Jakob disease in humans can wreck havoc on entire populations of livestock and wildlife.

A disease outbreak or natural disaster, depending on the location could cause a large scale catastrophic mortality event. In the case of a disease outbreak, animals within a certain radius would be culled and disposed (SAF 2005). There are many methods currently in place to dispose of mortalities that occur on a regular basis. These include incineration, composting, rendering, and burial (Engel et. al. 2004 & SAF 2005). In the occurrence of a large scale mortality event, each of these methods has its advantages and disadvantages. When dealing with carcass disposal due to a disease outbreak, many other factors such as disease transmission and the public health have to be taken into

consideration. In doing so, this may eliminate the use of some of the existing disposal practices.

Incineration is a practice commonly used for hundreds of years (Kastner & Phebus 2004). There are three main types of incineration techniques that have been employed to dispose of animal carcass. These include Open-air burning, fixed facility incineration and air curtain incineration (Kastner & Phebus 2004). Open air burning requires the use of fuels such as straw, diesel, coal, kindling wood and other combustible, low moisture content materials. Burning carcasses in open fields, pyres and other burning techniques not assisted by incineration equipment are encompassed by the process called open-air burning (Kastner & Phebus 2004). Fixed facility incineration is the basic concept behind crematoria. They include small scale on-farm incinerators, to crematoriums and large scale power plant types. From the start of bovine spongiform encephalopathy (BSE) outbreaks in the UK, carcasses along with rendered meat and bone meal (MBM) and tallow have been disposed by fixed facility incineration (Kastner & Phebus 2004). Fixed facility incineration is limited to its throughput (Kastner & Phebus 2004). Air curtain incineration uses a fan to force air through a manifold to speed up the incineration process. This incineration technique also needs wood and a burning fuel such as diesel. Each incineration technique poses a fire threat and by-product of ash. Each technique's capacity is determined by its throughput, the type of species and diseases that affected them (Kastner & Phebus 2004). Incineration is a disposal method not suitable for TSE's due to the stability of the prion unless they reached the required 850°C temperature for inactivation (SSC 2003a).

Composting animal carcasses is a common practice among livestock producers to dispose of their routine mortalities. Carcass composting is relatively less capital intensive than incineration and rendering, a better alternative to burial in areas with shallow water tables, and provides for quick removal and isolation of farm mortalities (Kalbasi et al. 2005). Composting a carcass involves placing it into a carbon rich environment. Decomposition of the animal in the environment by microorganisms produces water vapor, heat, carbon dioxide, and organic residue (compost). The compost can then be applied to agricultural land as a soil amendment. Composting animal carcasses can take 1 to 9 months before usable humus is ready. A small carcass is defined as less than 23 kg, a medium carcass up to 114 kg, a large carcass of 227 kg, or a very large carcass such as mature bovine and horses at 545 kg (Kalbasi et. al. 2005). It takes approximately 10, 90, 180 and 250 days respectively to finish their decomposition process (Kalbasi et. al. 2005). During the 2004 outbreak of Avian Influenza in British Columbia, composting was used to dispose of uninfected culled birds, while infected birds were first treated with biological heat treatment and then composted (Engel et al. 2004). Problems associated with using composting for a mass mortality event include maintaining proper temperature and moisture content, maintaining proper aeration, and creating an adequate carbon to nitrogen ratio (Fonstad et al. 2003).

Rendering livestock mortalities is the preferred method of disposal (SAF 2005). This method involves recycling carcasses into useful by-products such as meat and bone meal and tallow. In Western Canada, rendering plants are sparse and travel distances to the facilities make it less economical than other disposal options. Trucks are the main

method of transporting carcasses to the facilities. The trucks make many stops along the way; therefore have the potential to spread disease along their travels. In the event of a disease outbreak, rendering may no longer be an option for most producers due to the risk of contamination by the trucks. Rendering also destroys most pathogens if they are present due to temperatures (130°C) achieved during the process; although some diseases may not be destroyed by this process and would then deem rendering unacceptable (SAF 2005).

Burial is the specific study of this thesis. Livestock mortality burial is a practice that has been used both small scale and large scale (Engel et al. 2004). Burial of normally occurring mortalities is a common acceptable method used in Saskatchewan if managed properly (SAF 2005). Requirements for most burial pits include location in clay or till soil, in areas not subjected to flooding, and maintain 2-4 meters between the bottom of the burial pit and a usable water source (SAF 2005). In the event of a disease outbreak or natural disaster, when large quantities of carcasses need to be disposed of, burial has in the past been deemed a reasonable option depending on site conditions (Engel et al. 2004). During the 2001 foot and mouth outbreak in the UK, large scale burial sites were constructed to contain hundreds of thousands of mortalities (MacAtthur et al. 2002). Livestock mortality burial is a subject that has not been studied in great detail (Glanville 2000; CFIA 2006; Ritter and Chirnside 1995). With very few studies having been performed, it is concluded that there is a lack of scientific information on the subject (Freedman 2003). Burial seems logical to dispose of a large number of carcasses in a timely manner, but there is a lack of knowledge of the leachate composition that develops in the burial pits from the decomposing animals.

Determining the chemical composition of pure livestock mortality leachate has never been studied. One of the main goals of this thesis is to determine for bovine, swine and poultry the chemical composition of the leachate involved with their decomposition. From an environmental contamination standpoint, this is very important and will aid in determining the strength of the leachate and whether or not burial is still a viable option for mortality disposal.

Objectives of this research are to:

1. Determine the chemical composition of livestock mortality leachate for three species: bovine, swine and poultry
2. Develop simulations for contaminant transport using two different contaminant transport models MIGRATEv9 and Geo-Slope CTRAN
3. Based on the simulations, provide worst case scenario transport distances

Each of these objectives was addressed specifically through detailed field experiments and computer models.

2 LITERATURE REVIEW

Livestock burial sites, whether for normally occurring mortalities, or those built for a mass mortality event, pose a threat of contamination to soil and groundwater. Little is known about the leachate composition of a decomposing carcass and therefore little is known about the actual threat involved with a burial site (CFIA 2006). A few studies have been performed using experimental burial sites and monitoring the groundwater quality around them (Glanville 2000; Ritter & Chirnside 1995; Scudamore et al. 2002; MacArthur & Milne 2002; MacArthur et al. 2002; Environment Agency 2001).

Many early investigations focused on monitoring groundwater quality surrounding existing mortality disposal sites (Glanville 2000; Ritter & Chirnside 1995). These methods give a basic analysis of potential contaminant transport into groundwater systems. Most of these studies give a basis for a typical, on-farm, normally occurring mortality situation, not something as similar to a mass mortality event. Average livestock losses occurring on site are generally not considered a threat in most provinces and states; however, when dealing with a mass mortality event, the sheer number of culled animals to be disposed requires a more careful evaluation of disposal sites and the risk involved (Engel et al 2004; SAF 2005).

2.1 Previous Studies Involving Livestock Burial

Although livestock mortality burial has not been studied in great detail, a few case

studies exist (Glanville 2000; Ritter & Chirnside 1995; Scudamore et al. 2002; MacArthur & Milne 2002; MacArthur et al. 2002; Environment Agency 2001). The focus of the majority of the studies was to monitor groundwater quality around an existing or experimental burial site (Glanville 2000; Ritter & Chirnside 1995; Scudamore et al. 2001). None of the studies gathered and analyzed pure mortality leachate.

2.1.1 Livestock Burial Effect on Shallow Groundwater Quality

The Iowa Department of Natural Resources teamed with Dr. Thomas Glanville at the Iowa State University and performed two case studies to evaluate impacts in regards to the environment when dealing with livestock burial. The results of these studies concluded that the proper disposal of livestock mortalities can be more difficult than manure management due to the rapid breakdown of the animal carcass in the environment (Glanville 2000).

The first case study investigated turkey mortalities located in a poorly drained soil with seasonal high water tables between 0.3 to 0.9 meters (Glanville 2000). Two pits were constructed one year prior to the study containing 28,400 kg of Turkey Mortalities. The pits were at a depth of 1.8 m. To monitor groundwater quality, 9 monitoring wells were placed within 10 meters around the pit, and 3 others within 30 meters. These wells were used to monitor contaminant migration and background water quality. Groundwater samples were collected monthly for the first 15 months and again at 20 months and 40 months (Glanville 2000).

Glanville (2000) also studied burial trenches located in well drained, moderately permeable soil. The two trenches were excavated to 1.2 m deep and spaced 2.4 m apart. Six swine carcasses spaced evenly filled each trench. To obtain leachate samples, one of the trenches was lined with PVC sheeting and 100 mm of pea gravel, the other trench was not lined. Monitoring wells were placed within three meters around the trenches to monitor groundwater contamination (Glanville 2000). Leachate samples from the lined trench were obtained monthly from a PVC pipe buried vertically at one end of the trench and outfitted with a sump pump. The leachate was measured to examine the mass, concentration, and duration of decay products. Leachate collected was exposed to rainwater during the duration of the experiment and therefore is not a pure leachate example.

For both case studies, elevated levels of Biochemical Oxygen Demand (BOD) at 230mg/L, Ammonia-Nitrogen (NH_4-N) at 403 mg/L, Total Dissolved Solids (TDS) at 1527 mg/L, and Chloride (Cl) at 109mg/L were found in the sample sets. Although chloride concentrations were generally lower than the other contaminants, elevated chloride levels are generally the best indicator of burial-related groundwater contamination (Engel et. al. 2004). Localized contamination may persist for a decade or more in wet soil with a high seasonal water table and low groundwater flow velocity (Glanville 2000). Even in lightly loaded burial trenches constructed in well drained soil, complete decay may take two years or more (Glanville 2000). Neither case study demonstrated contamination more than a meter or two from the site, but this situation could easily be changed in areas with high water tables and high groundwater velocities.

2.1.2 Impact of Dead Bird Disposal on Groundwater Quality

To examine the impact of dead bird disposal on groundwater quality, Ritter and Chirnside (1995) performed a study on existing poultry disposal pits. To examine this impact, six existing disposal pits in Delaware were used and groundwater around these pits was monitored. The pits were still active, meaning that there was a normal loading rate of broiler mortalities of approximately 15-25 kg of dead birds per week. To monitor the potential pollution, two to three monitoring wells were placed around the pits at a depth of 4.5 meters. These pits were located in areas of sandy soils and very high water tables. The bottoms of many of the disposal pits are located in the groundwater for most or all of the year. By using this method of disposal, the potential for groundwater contamination is very high. Monitoring of these existing pits ran from 1987 to 1990 and samples were drawn every 4-8 weeks and analyzed for ammonia, nitrates, fecal coliforms and fecal streptococci.

This study concluded ammonia (NH_3-N) concentrations were high in two wells. Of the six total pits, three of them demonstrated an increase in ammonia concentrations at a lateral distance of 12 m. In some cases the ammonia concentrations reached 366 mg/l and streptococci concentrations reached 209 mg/100 ml. In all monitoring wells fecal coliform and fecal streptococcus concentrations were low and only detected 20% of the time during sampling (Ritter & Chirnside 1995). Bacterial contamination of groundwater by the disposal pits was low. They conclude that serious groundwater contamination may occur if a grower has a large number of birds killed and has to bury them (Ritter & Chirnside 1995).

2.1.3 Mass Burial Sites for Foot and Mouth Disease Outbreak in Scotland

The United Kingdom has had two major foot and mouth disease outbreaks, the first in 1967 and the second in 2001. The outbreak of 1967 was not nearly as significant as that of 2001 due to many reasons. Populations of livestock, especially sheep have greatly increased over the years to 40 million sheep and 11-12 million head of cattle (Scudamore et al. 2002). The outbreak of 2001 caused a cull of 1,281,271 livestock including cattle, sheep, pigs and goats (Scudamore et al. 2002). Disposal operations varied by location and included on-farm burial, on-farm pyres, rendering, incineration, mass burial sites and landfills. On-farm burial was limited in many areas due to the requirement of a Groundwater Authorization by the Environment Agency and the need to protect the environment (Environment Agency 2001).

Within a few weeks of the outbreak, other disposal options soon met their limits and the need for mass burial locations became apparent. At the early stages of designing the mass burial sites, rapid development of designs evolved and changed as construction took place. At the early stages, the pits were a single hole in the ground and by the end, pits were engineered with sophisticated liners and leachate collection systems which attempted to minimize the risk to groundwater (Scudamore et al. 2002). A total of seven sites were granted a Groundwater Authorization and became mass disposal sites. These included 5 sites in England, one in Scotland and one in Wales, with each site capable of multiple pits, each pit able to hold between 10,000 and 60,000 carcasses.

The most documented and used mass burial site was Birkshaw Forest located near

Lockerbie, Scotland. Approximately 440,000 culled animals were disposed at the site in six disposal pits. Three of the first pits constructed were unlined due to time constraints, but the remaining three were constructed with geo-synthetic clay liners (GCL) and all pits posses a leachate collection system (MacArthur & Milne 2002). The burial sites were commissioned in April 2001 and not capped until June 2001. During the first two months rainwater was allowed to enter the pits. To keep pollution at a minimum, leachate was collected and sent to a treatment plant.

Leachate arising from these disposal sites is very strong in comparison to other leachate found at landfills or livestock waste lagoons. Due to its complexity, leachate had to be collected and processed. Very soon after disposal, large quantities of high strength leachate were generated due to the release of body fluids and rain entering the pits (MacArthur & Milne 2002). Animal carcasses consist of proteins, fats and initially, urea. Proteins have a high nitrogen content of approximately 16%, and therefore it was anticipated that there would be a high ammoniacal-N concentration in the leachate (MacArthur & Milne 2002). Initially after one week, the chemical oxygen demand (COD) values started at approximately 160,000 mg/l and rapidly fell to around 20,000-40,000 mg/l with the ammoniacal-N rising to 2000-4000 mg/l (MacArthur et al. 2002). Sampling the leachate of the pits was very inconsistent due to exposure to heavy rains. Once capping was complete on each site, a more extensive leachate analysis was completed involving sampling monthly from July 2001 to April 2003 (MacArthur et al. 2002). The results of this leachate analysis can be seen in table 2.1.

Table 2.1 Leachate chemistry from burial sites in Scotland (from MacArthur et al. 2002)

Determinand	# of Samples	Maxima	Minima	Mean	Median	Standard Deviation
pH	199	8	5.9	6.9	6.9	-
Calcium mg/l	194	700	2	208	183	132
Iron mg/l	193	335	0	52	33	53
Phosphate as P mg/l	198	476	1	55	25	77
Ammonia mg/l	199	19200	28	3294	2700	2702
Total Oxidised Nitrogen	9	10	0.2	2.1	0.6	3.2
BOD	9	38500	300	12700	11600	12875
COD	199	134200	500	20414	16000	20216
Electrical Conductivity uS	194	45000	5	11210	10985	7435
Alkalinity as CaCO3	199	88200	152	11935	9400	10233
Suspended Solids	199	5432	10	389	260	486
Dry Residue	108	82400	470	6866	4960	8498

Pollution of the environment was monitored closely around all disposal sites. Impacts to streams and rivers were limited at 212 total incidents, with only one incident of a category 1 which cause major damage to the aquatic ecosystem and 11 category 2 incidents which cause significant damage to the aquatic ecosystem (Environment Agency 2001). Mass burial sites showed very few incidents. Most problems occurred with the leachate and were quickly resolved. Groundwater contamination is continuing to be monitored through boreholes at each site. These boreholes are located at varying distances and depths from the burial sites. No data is currently published from the borehole analysis. Water companies have found no contamination of public water supplies (Environment Agency 2001).

2.1.4 Conclusion of Livestock Burial Research to Date

The literature revolving around existing disposal pits does not provide evidence of initial

mortality leachate composition. Transport of leachate has been documented in studies by Thomas Glanville (2000) and Ritter & Chirnside (1995). It is important to understand the potential of the contaminant put into a burial site. By understanding the contaminant under investigation, proper burial guidelines can be determined.

2.2 Biochemical Composition of Mammalian Cells

Many elements are important constituents for the building blocks of life. Carbon, hydrogen, oxygen and nitrogen are the most essential elements found in living organisms. Out of every million atoms in the body, 993000 are made up of the major building block elements: carbon, hydrogen, oxygen and nitrogen (Moore et al. 2002). Table 2.2 shows the abundance of elements found in the body in atoms per million atoms. The table includes the building block elements along with all other major minerals found in the body. There are also trace elements found in the body that only occur in very minute amounts.

Table 2.2 Elements found in the human body (from Moore et al. 2002)

Element	Atoms/Million Atoms in the body
Hydrogen	630000
Oxygen	255000
Carbon	94500
Nitrogen	13500
Calcium	3100
Phosphorus	2200
Chlorine	570
Sulphur	490
Sodium	410
Potassium	260
Magnesium	130

Hydrogen is one of the four main building blocks of all cells. It is mostly present in the body in the form of water. Water makes up approximately 70% of the body. H_2 can

also be found in small amounts in cells.

Oxygen atoms are present in water and water is essential to life. O_2 gas is also important during respiration for living organisms. Oxygen is a component of major biomolecules that make up the body in structures such as carbohydrates, proteins, fats.

Carbon is defined as the building block of life. It is found in many forms in the body and helps compose many structures responsible for existence. Carbon is mostly found in the form of bicarbonate (HCO^-_3) and carbonate (CO^{2-}_3).

Nitrogen composes 80% of the air we breathe, and is found in the body in many forms. One of the most common is cyclic nitrogen compound forming a DNA strand along with a sugar unit and phosphate (Moore et al. 2002). Nitrogen is also a key component in the structure of amino acids and nucleic acids that form proteins in the body. Crude protein makes up approximately 15% of an adult human body (Mitchell et al. 1945). The amino group in most proteins is formed with an H_2N compound (Mitchell et al. 1945). Nitrogen also helps form the peptide linkages of amino acid groups covalently bonded to each other (Moore et al. 2002).

Of the remaining elements found in the body, calcium and phosphorus make up approximately 2.2% of the total body mass. Through various research, it has been determined that calcium makes up an estimated 1.3 to 3.6 percent of the body and phosphorus 0.63 to 1.2 percent (Mitchell et al. 1945). The skeleton and teeth comprise 99% of the total calcium and four fifths the total phosphorus in the body (Sherman et al.

1926). The three main ionic forms of phosphorus found in the body include PO_4^{3-}, HPO_4^{2-} and $H_2PO_4^-$ (Moore et al. 2002). The phosphate ion is the main building block of bones and teeth based on calcium hydroxyphosphate – apatite, for adenosine triphosphate (ATP) and for the phosphate backbone of RNA and DNA as well as other biochemical molecules (Childs 2001). Calcium not only helps form stable bone material, it also is a key factor in the formation of cell walls and is important for blood clotting.

Other elements such as potassium, chlorine, sodium, sulphur and iron all have important functions in the body. Potassium is the main cation inside cells and is essential for heart and nerve functions. Sodium is the main extracellular cation and is also important for nerve function. Chlorine is mainly present in the form of hydrochloric acid in the stomach for digestion. Sulphur is present in fats, body fluids, skeletal minerals and is also a key component in determining a protein's tertiary structure (Moore et al. 2002). Iron is the key component of aerobic activity inside the body and is present in very small amounts. It is located in the haemoglobin and is responsible for carrying oxygen to the body.

Along with the major and minor elements in the body, small amounts of trace elements such as Copper, Cobalt, Mercury, Arsenic etc. also exist. Some of them have a small function in the body; others have not yet had a function discovered. These elements are present in amounts so small, that for mortality leachate, it is expected that they will not be detectable.

2.3 Groundwater Pollutants and Standards

The quality of potable water is determined by the amount of total dissolved solids (TDS). Pure water is made up only of Hydrogen and Oxygen ions. Water in the environment is never found in its pure state. Groundwater contains many dissolved constituents including microorganisms, gases, inorganic and organic materials (Environment Canada 2004). The depth of soils and subsurface geology through which the water passes through has the greatest effect on the chemicals dissolved in natural groundwater. Table 2.3 shows water quality standards that are not to be exceeded in Canada, it also compares them to the WHO, EU, and SDWF standards.

Table 2.3 Drinking water quality standards (from Environment Canada 2004)

Dissolved inorganic compounds		
Alkalinity	500 (SK)	mg/L
Aluminum	0.2 (WHO)	mg/L
Ammonia	50 (EU)	mg/L
Arsenic	0.025	mg/L
Boron	5	mg/L
Cadmium	0.005	mg/L
Calcium	100 (EU)	mg/L
Chloride	250	mg/L
Chlorine (total residual in chlorine-disinfected water)	>0.5	mg/L
Chlorine (free residual)	>0.1	mg/L
Copper	1	mg/L
Fluoride	1.5	mg/L
Iron	0.3	mg/L
Hardness	800 (SK)	mg/L
Lead	0.01	mg/L
Magnesium	200	mg/L
Manganese	0.05	mg/L
Mercury	0.001	mg/L
Nitrate (as nitrate-nitrogen)	10	mg/L
Nitrate (as nitrate)	45	mg/L
Selenium	0.01	mg/L
Sodium	200 (SK 300)	mg/L
Sulphate	500	mg/L
Sulphide	0.05	mg/L
Total dissolved solids (indicator of the amount of salt in solution)	500 (SK 1.500)	mg/L
Zinc	5	mg/L

Note: (EU) signifies European Union Standards, (SK) represents Saskatchewan Standards, (WHO) represents World Health Organization Standards

The contamination of groundwater occurs from three basic mechanisms: by natural processes, by man's disposal practices and by spills, leaks and agricultural activities (Environment Canada 2004). One factor that makes this such a serious problem is its long-term effects. Most contamination sites are small, but the severity lies in the fact

that the extreme sites have been contaminating for a long period of time. Refining practices from the mid 1800's to 1900's dumped very toxic wastes such as tar, treated sewage, waste fuels and solvents into rivers, high infiltration lagoons and burial disposal sites. Today, contaminates are classified to originate from either point sources or distributed sources.

Landfills and earthen manure storages are documented leachate contaminants that could be similar to that of livestock mortality leachate. Examples of concentrations of earthen manure storages can be taken from Fonstad (2004) and are shown in table 2.4. Those values are compared to landfill leachate characterized from Freeze and Cherry (1979) and Thornton et al. (2001) also shown in table 2.4.

Table 2.4 Landfill and Earthen Manure Storage Leachate Characteristics

		Landfill Freeze & Cherry (1979)	Landfill Thornton et al. (2001)	Manure Storage Fonstad (2004)
pH		4-8	7.2	7.6
Alkalinity	mg/l	500-10000	4600	11701
Chloride	mg/l	300-3000	1965	1351
Sulphate	mg/l	10-1000	52	699
Calcium	mg/l	100-3000	125	190
Magnesium	mg/l	100-1500	135	96.2
Sodium	mg/l	200-1200	1340	734
Potassium	mg/l	200-1000	490	2023
Ammonium	mg/l	10-1000	1029	3879
Nitrate	mg/l	0.1-10		3.6
Iron	mg/l	1-1000	13.9	3.28

2.4 Methods of Contaminant Transport

When a contaminant is introduced at or below the soil surface, such as livestock burial leachate, there are many pathways it can take before reaching the saturated zone and

being transported through an aquifer system. The transportation and fate of the contaminant is partially regulated by the chemistry of that contaminant. There are two main groups of properties that determine the transport of the chemicals: physiochemical properties such as solubility, vapor pressure, partition coefficient, sorption-desorption, volatilization; and chemical transformations such as oxidation-reduction behavior, hydrolysis, halogenation-dehalogenation, and photochemical breakdown (Palmer 1996). Solubility is of key importance due to each chemical's ability to dissolve in water. Once the substance is dissolved in water, its mobility can be determined and can be grouped as follows:

Table 2.5: Mobility of Chemicals (Rail 1989)

Extremely Mobile	Chlorides & Sulfates
Somewhat Mobile	Calcium, Magnesium & Sodium
Generally Mobile	Silicon, Phosphorus & Potassium
Slightly Mobile	Iron, Chromium, Aluminum & Lead

2.4.1 Mass Transport through the Saturated Zone

Transport of chemicals through the saturated zone is mainly determined by the ion exchange capacity of the soil particles, the porosity of the soil and degree of saturation (Fetter 1999). The presence of clays and silts increases the ion exchange capacity. The ion exchange capacity is the sum of cation exchange capacity (CEC) and anion exchange capacity (AEC). CEC is determined by the electrostatic double layer on a clay particle. The cations associated with this layer can be replaced by other cations in solution (Fetter 1999). Although CEC is the most dominant in the uptake of ions, some colloids attract anions. Due to the structure of clay colloids, there is preference in the exchange of cations and that order is shown below (Fetter 1999).

$$Al^{3+} >> Ca^{2+} > Mg^{2+} >> NH_4^+ > K^+ > H_3O^+ > Na^+ > Li^+$$

However, if a soil is flooded with a solution containing a large concentration of one cation, the normal cation exchange order can be reversed (Fetter 1999). The order can also be changed in this regard by activity coefficients (Fonstad 2004).

The exchange of anions and cations occurs as solution passes through the unsaturated zone in the form of mass transport. There are many equations derived by different soil scientists that describe this phenomenon. One such equation is known as Fick's Law which describes the steady state diffusion of a solute in the soil and is as follows (Fetter 1999):

$$J = -D_s^*(\theta)\frac{dC}{dz} \qquad [1]$$

where:

J = *mass flux of solute per unit area per unit time*
$D_s^*(\theta)$ = *soil diffusion coefficient*
dC/dz = *concentration gradient in the soil moisture*

Van Genuchten (1980) has formed a more popular mass transport equation dealing with equilibrium solutions stating that the solute can be in either a dissolved form or absorbed by the soil, the relationship between the concentration in solution and the absorbed concentrations need to be known. Van Genuchten (1980) has derived this equation for many different boundary conditions; one so derivation used commonly is as follows:

$$C(z) = \frac{\xi}{\eta} + \left(C_o - \frac{\xi}{\eta} \right) \left(\frac{2v}{v+u} \right) \exp\left(\frac{(v-u)z}{2D_s} \right) \qquad [2]$$

where:

$C(z) = concentration\ in\ solution$

$C_o = initial\ solute\ concentration$

$\xi = zero-order\ rate\ terms\ for\ production$

$\eta = rate\ const.\ for\ 1st\ order\ decay\ in\ liquid\ and\ solid\ phases$

$v = pore\ water\ velocity$

$u = v\left(1 + \frac{4\eta D_s}{v^2} \right)^{1/2}$

$D_s = soil\ moisture\ diffusion\ coefficient$

$z = position\ in\ soil\ column$

Preferential flow paths are important in contaminant transport and should be carefully considered. Cracks, gaps and animal tunnels or holes are all considered preferential flow paths and each can form a rapid route for transport. When contaminants follow preferential flow paths, there is less time for ion exchange and therefore can be more harmful to zones beneath them.

2.4.2 Ion Exchange in the Saturated Zone

The negative charges on soil particles are important due to their abilities for cation exchange. In the saturated zone, groundwater is assessed as a partial equilibrium system (Fetter 1999). Equilibrated reactions form an equilibrium constant (K). For the following reaction the equilibrium constant is found by:

$$\mathbf{aA + bB = cC + dD}$$

$$K = \frac{(C)^c (D)^d}{(A)^a (B)^b} \qquad [3]$$

where:

a, b, c, d = *respective number of moles involved*

A, B, C, D = *molar or molal concentrations*

The above equations are normally only valid for very diluted solutions. When dealing with non-dilute solutions such as the livestock burial leachate from burial pits, ions interact electrostatically with each other. Once this begins to happen, an adjusted molar concentration to an effective concentration needs to be determined by establishing an activity coefficient (γ). There are many forms of solving for activity coefficients, including the Debije-Huckel equation, the Extended Debije-Huckel equation, the Davis Equation, and the Pitzer Equation. A simple rule to remember is that activities are usually smaller for multivalent ions than ions with a single charge. To determine whether or not a mineral is in equilibrium with its solution, the saturation index can be determined. For an equilibrium situation the saturation index will be zero, for undersaturation less than zero and for supersaturation greater than zero. This determines whether an element is being precipitated or dissolved in solution.

2.4.3 Transport by Concentration Gradients

Diffusion is one common way in which solutes are transported through the saturated zone. Diffusion is defined as solutes moving from an area of greater concentration toward an area of lower concentration. This model can be used for fluids in motion or for stagnant situations, as long as a concentration gradient exists. Diffusion is the predominant mechanism of transport only in low permeability hydrogeologic regimes

(Fetter 1999). Fick's first law describes diffusion in one dimension as (Appelo 1996):

$$F = -D\frac{dC}{dx} \qquad [4]$$

where:

$F = mass\ flux\ of\ solute\ per\ unit\ area\ per\ unit\ time$

$D = diffusion\ coefficient\ (L^2 / T)$

$dC / dx = concentration\ gradient\ (M / L^3 / L)$

Fick's second law describes the diffusion situation where concentrations are changing with time and is stated as:

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial x^2} \qquad [5]$$

2.4.4 Transport by Advection

Solutes are also carried along the flow paths with flowing water by a process called advective transport. The solute concentration amounts are a function of the concentrations present in the groundwater and the amount of groundwater that is flowing. To determine water flow, the average linear velocity (v) is used. Advection is described as plug flow, all the pore fluid is being replaced by the invading solute front (Fetter 1999). The equation used to calculate one-dimensional advective transport is defined as (Fetter 1999):

$$\frac{\partial C}{\partial t} = -v_x \frac{\partial C}{\partial x} \qquad [6]$$

where:

v = *average linear velocity*

dC / dx = *concentration gradient* $(M / L^3 / L)$

The method of advection will transport solutes at different rates throughout different types of porous media. It is difficult to use only the advection method to determine the resulting distribution of solute.

2.4.5 Advection-Dispersion/Diffusion

The previous methods are independently derived, but for a more reasonable calculation, they are combined into a one dimensional mass flux equation:

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial x^2} - v_x \frac{\partial C}{\partial x} \qquad [7]$$

This equation is based on hydrodynamic dispersion which is the sum of mechanical dispersion and diffusion. To evaluate which method is contributing the most, a Peclet number is determined. The Peclet number is defined as

$$Pe = \frac{v_x d}{D} \qquad [8]$$

where:

v = *pore water velocity*

D = *diffusion coefficient in pure water* (m^2 / s)

d = *diameter of particles*

All variables have been defined above. At low Peclet numbers, diffusion dominates; at high Peclet numbers, advection dominates the transport process. The advection/dispersion equation can be expanded to two or three dimensions.

2.5 Modeling of Contaminant Transport

Many computer models exist to evaluate the transport of contaminants into the underlying soil layers and groundwater systems. This thesis will focus on using two of the existing modeling software packages to predict transport of mortality leachate beneath burial sites.

2.5.1 MIGRATE v9

MIGRATEv9 (Rowe et al. 1995) is a contaminant transport model based on the two-dimensional solution to the advection-dispersion equation. Two dimensional contaminant transport for a single solute and for a layered deposit of finite thickness is the basis of development of the analytical solution (Rowe et al. 1995). To find the solution to the two-dimensional equation, a Laplace transform and a Fourier transform are used. Because this program uses a time-marching scheme, it is different than most finite element and finite difference models in that it can determine the concentration at a time step without calculating concentrations at earlier times (Rowe et al. 1995).

2.5.1.1 Model Assumptions

MIGRATEv9 (Rowe et al. 1995), allows each layer in the model to have different properties. By using a finite layer technique, the model assumes that in any layer, the parameters do not vary with lateral position. The model also assumes that Darcy velocities also do not vary laterally and for models where velocities need to vary laterally, MIGRATEv9 may not be suitable (Rowe et al. 1995). As with any model, judgment is required to determine whether or not accuracy is being achieved.

2.5.1.2 Integration Technique

The solution is found using a Fourier transform involving numerical integration of the governing equations. Gauss quadrature is used to achieve this by numerically performing the integrations and decomposing the integration range into subintervals (Rowe et al. 1995). Accuracy of the model is gained by geometries and properties of the model under investigation. Integration levels can also be changed in the Gauss quadrature into four levels, FINE, NORMAL, WIDE and SELECT (Rowe et al. 1995). By changing the integration level, model computation time can be increased or decreased depending on the selection. The error involved with the integration technique depends upon the problem and the depth of the point of interest beneath the source (Rowe et al. 1995). Errors seem to be greater at the surface outside the landfill, and decreases with depth below the landfill (Rowe et al. 1995). It is also noted that by using a finite layer technique, there will be no discretization errors associated with the model. Furthermore, problems associated with oscillation of solutions and the selection of time steps in a finite element or finite difference technique does not appear with the finite layer technique (Rowe 1990).

2.5.1.3 Boundary Conditions

The governing two-dimensional, finite layer, contaminant transport function in a saturated environment MIGRATEv9 employs is as follows (Rowe et al. 1995):

$$n\frac{\partial c}{\partial t} = nD_{xx}\frac{\partial^2 c}{\partial x^2} + nD_{zz}\frac{\partial^2 c}{\partial z^2} - nv_x\frac{\partial c}{\partial x} - nv_z\frac{\partial c}{\partial z} - \rho K_d\frac{\partial c}{\partial t} - n\lambda c \qquad [9]$$

where:

c = *concentration of contaminant at depth z at time t*

D = *coefficient of hydrodynamic dispersion at depth z*

v = *groundwater (seepage) velocity at depth z*

n = *porosity of the soil at depth z*

K_d = *distribution/partitioning (sorption) coefficient at depth z*

$v_a = nv$ = *Darcy velocity*

λ = *decay constant of the contaminant species*

Solving equation 9 involves the use of boundary conditions specific to the problem at hand. Boundary conditions can be set at the top and bottom of the soil deposit being modelled. The top boundary condition is the point at which the contaminant is introduced to the soil deposit. There are four top boundary conditions offered by MIGRATE, these are constant surface concentration, finite mass surface concentration, impermeable surface with a buried source or a zero concentration surface with a buried source (Rowe et al 1995). The bottom boundary conditions may either be zero flux, zero concentration or an aquifer.

Other parameters that are input into the model include source characteristics such as landfill base width, surface width and concentration of the source. For each layer in the model, specific soil characteristics can be entered. Changes can be made to layer thickness, dry density, porosity, diffusion coefficients, Darcy velocities, distribution coefficient, half-life and sink removal rates. Sub layers in each layer are used in the output of the calculated concentrations with depth (Rowe et al. 1995). Dimensional units can be selected in the model when inputting parameters.

2.5.2 Geo-Slope 2004 CTRAN

Geo-Slope 2004 CTRAN is also a contaminant transport model based on the solution to

the two dimensional advection-dispersion equation (Krahn 2004). CTRAN is based on the finite element technique which is different than that of MIGRATE's time marching, finite layer calculation scheme. CTRAN uses a Galerkin approach to derive the solution to the finite element equation (Krahn 2004). The finite element technique employs discretization of the area under investigation into elements. Each element is connected by nodes. The nodes are used for calculating the solution to the finite element equation by using integration techniques.

2.5.2.1 Integration Technique

Concentrations anywhere in the element can be expressed in terms of the nodal concentration

$$C = < N > \{C\} \qquad [10]$$

where:

<N> = a matrix of interpolating (or shape) functions

{C} = the vector of nodal concentrations

Using this approach the residual function of the governing differential equation is:

$$R(c) = \Theta D \frac{\partial^2 C}{\partial x^2} - U \frac{\partial C}{\partial x} - \lambda S \rho_d - \left(\Theta + \rho_d \frac{\partial S}{\partial C} \right) \frac{\partial C}{\partial t} \qquad [11]$$

where:

Θ = *volumetric water content*

D = *hydrodynamic dispersion coefficient*

U = *Darcy velocity*

S = *adsorption*

λ = *decay coefficient*

ρ_d = *bulk density of the soil*

The solution to the finite element equation is a function of time and the integration is performed by a finite difference approximation scheme (Krahn 2004). The user can either use a Backward Difference Approximation, or a Central Difference Approximation. Unlike MIGRATE, to solve for nodal concentrations at a certain time increment, CTRAN needs to know the nodal concentrations at the start of the time increment, therefore the initial conditions of the problem must be known for a solution to the transport equation (Krahn 2004).

2.5.2.2 Boundary Conditions

To solve equation 11, CTRAN uses boundary conditions to specify parameters in the equation. In basic form, the software is computing for either an initial concentration or flux, and to do this, the model must have a concentration or flux specified at various nodes. The solution can only be obtained if at least one node in the finite element mesh has a specified concentration or flux value (Krahn 2004).

CTRAN setup involves determining the model size and material properties. An initial finite mesh is created and parameters such as soil type, water content, Darcy velocity, porosity, diffusion, decay etc. have to be entered into the model to obtain a solution. CTRAN is a more complex model than MIGRATEv9 as it allows boundary conditions to be specified not only at the top and bottom of the model, but anywhere else as well and can be varied throughout the model.

2.6 Leachate Speciation by PHREEQC

To determine the presence of element species such as phosphorus and ammonia in the

mortality leachate, PHREEQC will be used. PHREEQC computes aqueous speciation calculations by inputting a chemical composition for a solution. By inputting chemical composition of the solution, PHREEQC then performs mole balances for elements or element valence states, activity of water and ionic strength (Parkhurst & Appelo 1999). The output of the speciation shows results for the distribution of species and saturation indices.

3 METHODS AND MATERIALS

3.1 Leachate Chemistry

Leachate arising from a livestock burial site has not been accurately characterized. To characterize the chemical composition of this leachate, five burial sites were constructed south of Saskatoon, Saskatchewan. It was also deemed imperative to determine whether or not different species of livestock created a different leachate composition. Three common species of livestock were selected for the study and include bovine, swine and poultry. Each species was analyzed separately to determine its leachate composition.

3.1.1 Burial Site Construction

A total of five livestock mortality pits were constructed for determination of mortality leachate composition. Each pit was excavated to dimensions of 7 x 9 meters and a depth of 2.5 meters. A buffer zone of 20 meters was left between each pit. The pits were lined with 40 mil polyethylene and a collection system was located at the bottom center of each pit. (Fig. 3.1)

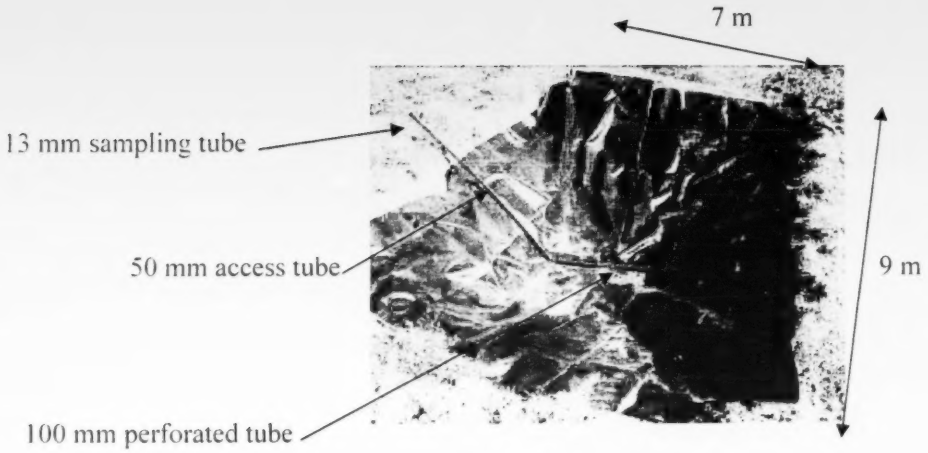

Figure 3.1 Lined burial pit

Three of the five pits (one of each species) were completely sealed by placing a 40 mil polyethylene cover above the carcass to allow an pure leachate sample. This cover also contained two vent 50 mm ABS pipes to allow for gas transfer to keep the pits stable.

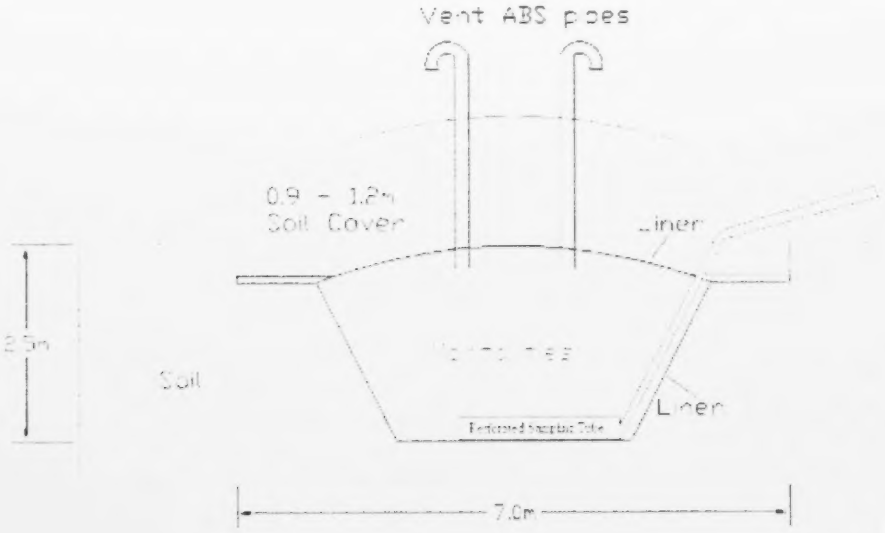

Figure 3.2 Burial pit cross section completely sealed

The remaining two pits (bovine and swine) possessed the same 40 mil polyethylene liner and leachate collection system in the bottom of the pit, but they did not have a top

polyethylene cover. These two pits were topped with soil excavated from the pit. (Fig. 3.3) One of these pits contained bovine, and the other swine. To control water infiltration and varmint intrusion, all five pits were capped and mounded with approximately 0.9 to 1.2 meters of soil. Due to settlement issues, soil was added or reshaped on the mounds a few months after burial.

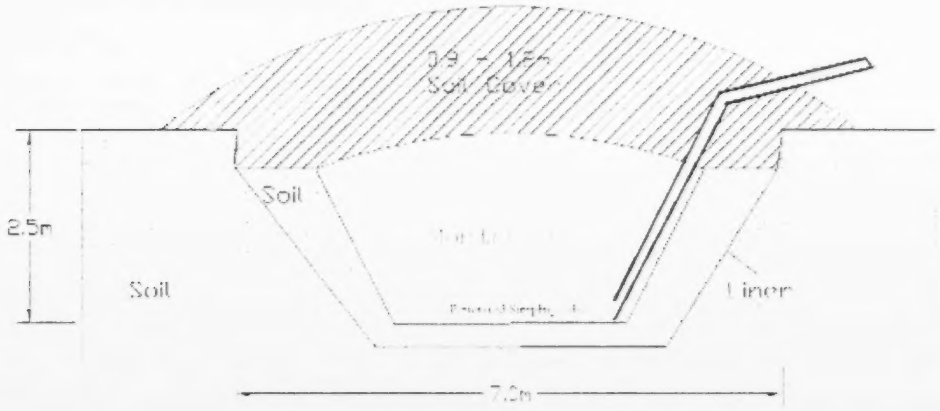

Figure 3.3 Burial pit cross section with soil cover

Pit No. 1 (Poultry) was installed August 4, 2005 and contained 1300 kg of chicken carcass in an environment sealed from contact with soil and water. Pit No. 2 (Swine) was installed August 9, 2005 and contained approximately 5900 kg of swine carcass in a sealed environment. Pit No. 3 (Swine); also installed August 9, 2005 contained approximately 5900 kg of swine carcass layered with in situ fine sandy clay soil. Pit No. 4 (Bovine), installed August 9, 2005 contained approximately 3920 kg of bovine carcass in a sealed environment. Pit No. 5 (Bovine), installed August 17, 2005 contained approximately 9750 kg of bovine carcass layered with in situ fine sandy clay soil.

3.1.2 Leachate Collection System

Each of the five pits contained a leachate collection system. This system consisted of a 100 mm x 1 m perforated polyethylene pipe covered by a filter sock placed in the bottom center of the pit on top of the polyethylene liner. The 100 mm perforated pipe was attached to a 50 mm ABS conduit pipe that ran to the surface. Inside this system of pipes were two 13 mm polyethylene lines. The ends of these lines were located in the perforated pipe and each end was separated by a distance of 600 mm (Fig. 3.4). This separation distance allowed for leachate mixing directly prior to sampling.

Figure 3.4 Leachate collection system

3.1.3 Sampling Methods

Sampling was conducted periodically over a period of fourteen months. Periodic samples were taken to observe the trends in chemical composition as decomposition occurred. Leachate samples were drawn from the pits at two weeks post burial (August 17, 2005) for Pit No. 1 (poultry), and at two weeks for the remaining pits on August 25, 2005. Samples were then taken on September 21, 2005, October 20, 2005, November

23, 2005, May 25, 2006 and October 26, 2006. Due to winter conditions, samples were not taken between November 2005 and May 2006.

Sampling was achieved by the use of a peristaltic pump connected to each 13 mm line at the ground surface. The purpose of the lines was to provide a mechanism of mixing prior to collecting the leachate sample. The peristaltic pump was used to circulate leachate fluid through the lines for five minutes. (Fig. 3.5) Once leachate was thoroughly mixed within the perforated chamber, a 1 litre sample was drawn and stored at 4°C during transport.

Figure 3.5 Leachate sampling

Samples were taken to SRC Analytical Laboratories in Saskatoon for analysis. Properties analyzed include: Ammonium by colorimetry, Bicarbonate and Carbonate by alkalinity, Chloride by colorimetry, Hydroxide, pH, Specific Conductivity, Alkalinity by ICP-AES, Nitrite-Nitrate Nitrogen by colorimetry, Total Kjeldahl Nitrogen, Inorganic and Organic Carbon with the Dohrmann Phoenix 8000 Carbon Analyzer, and

Aluminum, Barium, Beryllium, Boron, Cadmium, Calcium, Chromium, Cobalt, Copper, Iron, Lead, Magnesium, Manganese, Molybdenum, Nickel, Phosphorus, Potassium, Silicon, Silver, Sodium, Strontium, Sulfate, Sulfur, Titanium, Vanadium, Zinc and Zirconium by ICP-AES. Analytical methods and detection limits are given in Appendix A.

3.2 Modeling Methods

In order to determine the potential impact mortality leachate can have on soil and groundwater systems, contaminant transport models were created. To generalize the model, it was assumed that each model be a worst case scenario at each parameter. To determine the validity of the model, it was created in two separate software packages, MIGRATEv9 and Geo-Slope CTRAN 2004. Each model is a two dimensional model dealing with vertical and horizontal contaminant transport.

3.2.1 Geometry and Boundary Conditions

In a meeting with Saskatchewan Agriculture and Food, it was determined that the burial method that would be used for the province in an emergency situation would be a trench 2 meters wide by 4 meters in depth (Jansen 2006). These dimensions were used to create the computerized model. To determine the amount of leachate capable of transport, the following was assumed and based on an average bovine. The average weight of a bovine is 545 kg, in which seventy percent of is water (CFIA 2006). The Canadian Food Inspection Agency (CFIA) has determined that 1.5 m^3 is required for burial space of one animal (2006). It was also assumed that a decomposing/decomposed carcass was approximately fifty percent porosity. Using these assumptions, the amount

of free fluid available for transport can be calculated. This assumption equates to approximately 1 m^3 of leachate available for transport per meter of trench.

3.2.1.1 MIGRATEv9

The MIGRATEv9 model was created to model downward advective movement of the leachate. The model also included vertical and horizontal diffusion. The geometry and boundary conditions for this model are shown in Figure 3.6.

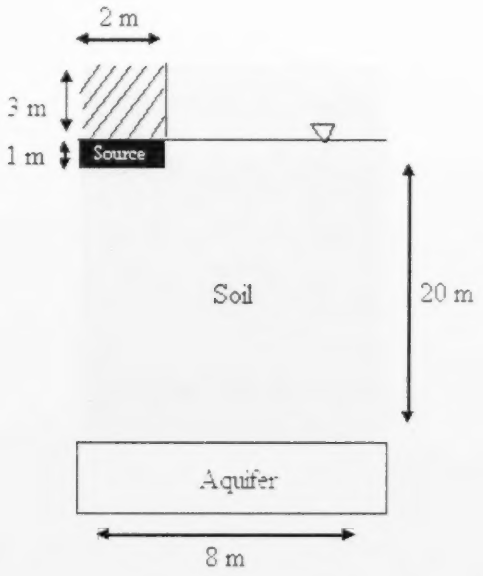

Figure 3.6 MIGRATE model dimensions

The top boundary condition for the model was a finite mass source with a depth of 1 meter. To simulate a worst case scenario, the bottom of the pit is located at the water table and a depth from the source to a simulated aquifer was 20 meters. The model also calculated concentrations at horizontal distances up to 8 meters. The concentration at the source is assumed to be equal to 1.0 so that all predictions are for a relative

concentration. The transport of the contaminant is conservative, that is, it is not attenuated or absorbed onto the soil particles.

3.2.1.2 Geo-Slope CTRAN

The Geo-Slope 2004 model initial setup is shown in Figure 3.7 with dimensions in meters. The top boundary conditions for this model include a finite mass source located at the nodes of the red triangles and include a contaminant depth of 1 meter at 15000 mg/l. The pit bottom is level with the water table to simulate a worst case scenario. The model was also allowed to calculate horizontal concentrations for 5 meters on either side of the pit. The contaminant in this model is also considered conservative.

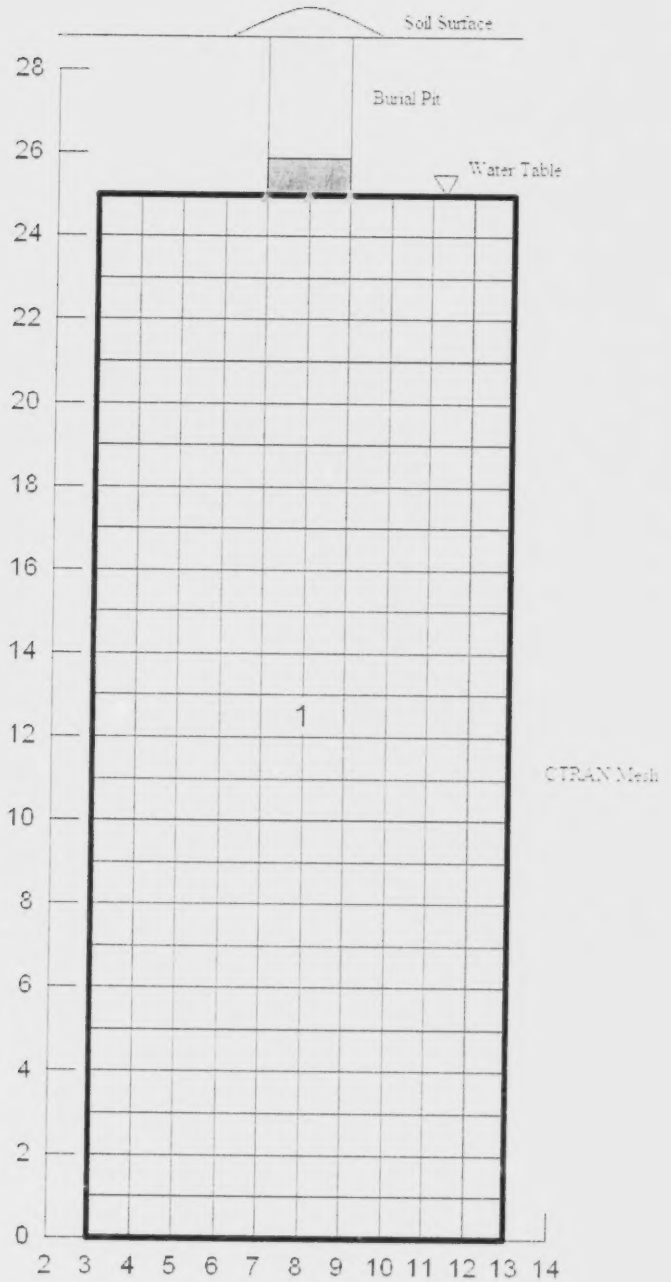

Figure 3.7 Geo-Slope CTRAN model dimensions

3.2.2 Material Properties

In order to determine the potential impact in different soil settings, different situations of soil systems needed to be considered. To predict groundwater flow and solute transport, the following material properties are needed: Darcy flux, coefficient of molecular diffusion, dispersivity and matrix porosity. The models should be considered over a reasonable range of all material parameters. The same material parameters were used in both MIGRATEv9 and Geo-Slope CTRAN 2004.

3.2.2.1 Porosity

Porosity for different geologic materials vary over a small range, with sands and gravels between 0.2 and 0.5, silts between 0.3 and 0.5 and clays between 0.3 to 0.7 (Freeze & Cherry 1979). To generalize the models, three porosities were selected to provide a range from sands to clays. These values are 0.2, 0.3 and 0.4 in the MIGRATEv9 models and standardized at 0.3 for all Geo-Slope CTRAN 2004 models.

3.2.2.2 Coefficient of Molecular Diffusion

The coefficient of molecular diffusion can be defined as the coefficient of molecular diffusion for the species in free water multiplied by the tortuosity factor. The tortuosity factor represents a reduction in diffusion due to the presence of a porous medium (Freeze and Cherry 1979). The coefficient of molecular diffusion varies from 1 x 10-10 to 2 x 10-10 m^2/s at 25°C for most salts. Temperature can reduce this value by 50 percent if the temperature is reduced from 25°C to 5°C. The tortuosity factor can range typically from 0.5 to 0.01 (Freeze and Cherry 1979). From this the following molecular diffusion coefficients were used for the MIGRATE model: 0.003, 0.01 and 0.03 m^2/year.

A value of 0.03 m^2/y was used in the Geo-Slope CTRAN 2004 model.

3.2.2.3 Coefficient of Mechanical Dispersion

Dispersion is the mixing of solutes caused solely by the motion of fluid. In the direction of groundwater flow it is found by the product of velocity and a mixing length. It is typically estimated at 1/100 of the travel distance (Gelhar et al. 1992). It is impossible to measure dispersivity at any site without performing a tracer test. From this, a value was calculated for each model depending on available transport distance.

3.2.2.4 Darcy Velocity

The horizontal and vertical Darcy velocities are a function of porosity and groundwater velocity. For the models, velocity was only used in the vertical gradient. Groundwater velocities ranging from $1x10^{-10}$ m/s, $1x10^{-9}$ m/s and $1x10^{-8}$ m/s were used in the MIGRATE and Geo-Slope models to demonstrate typical velocities for a clay, till and sand soil system (Freeze and Cherry 1979, Hendry 1982, Keller et al. 1986).

3.2.3 Model Duration

To predict the duration of contaminant transport, the simulations in MIGRATEv9 and CTRAN 2004 produced results at ten, twenty, fifty and one hundred years. From these results, plots were created to quantify the difference in transport between each simulated soil type.

3.3 Leachate Speciation Model

To provide a representation of the element species that can be present in the mortality leachate, an average value from all three species bovine, swine and poultry was used.

Table 3.1 shows the values entered into PHREEQC for the speciation calculations. The database used to calculate the speciation comes from Parkhurst and Appelo (1999) and is titled phreeqc.dat.

Table 3.1 PHREEQC Speciation Initial Concentration Values

Element	Average Concentration (mg/l)
pH	6.7
Ammonia	9640
K	2370
Na	1910
Ca	55
Mg	54
Alkalinity	31805
Cl	2658
Sulphate	4360
P	1584
Fe	36
Si	21

4 RESULTS AND DISCUSSION

4.1 Livestock Decomposition Leachate Chemistry Results

The major elements found in livestock mortality leachate include ammonium, bicarbonate, TKN, calcium, magnesium, iron, potassium, phosphorus, sulphate, sodium, chloride, along with other trace amounts of other elements. The major elements found in the burial leachate will be discussed in the next section. Nitrogen found in ammonium and TKN are of the most concern due to their high values ranging from 2400 to 15200 mg/l and 11000 to 20200 mg/l respectively. High levels of phosphorus ranging from 920 to 2090 mg/l and bicarbonate from 25700 to 53400 mg/l was also discovered. Tables 4.1 – 4.5 show the analytical results for each burial pit per sampling date. Pit No. 3 & 5 also contained soil mixed in with the mortalities.

Table 4.1 Poultry (Pit No. 1) Leachate Chemistry

DATE of SAMPLE		Aug.17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/l	25700	33200	37000	36000	38800	39000	39600
Carbonate	mg/l	0	0	0	0	0	0	0
Chloride	mg/l	3480	2860	2540	2470	2690	2670	2360
Hydroxide	mg/l	0	0	0	0	0	0	0
pH	pH Units	6.23	6.2	6.4	6.42	6.44	6.48	6.52
Specific Conductivity	uS/cm	45900	30700	37400	37600	36800	36900	45200
Total Alkalinity	mg/l	21100	27200	30300	29500	31800	3200	32500
Ammonia as Nitrogen	mg/l	4500	4542	9860	11200	8370	12700	10100
Nitrate + Nitrite nitrogen	mg/l	0	9.1	3.7	1.8	0	2	4.9
Total Kjeldahl Nitrogen	mg/l	19500	16500	17800	20200	18100	17700	17300
Inorganic Carbon	mg/l	5060	6530	7280	7080	7630	7670	7790
Organic Carbon	mg/l	47000	55000	55000	62000	58000	78000	99000
Aluminum	mg/l	1.5	1.4	0	0.8	0.6	1	0
Barium	mg/l	0.6	5	0.7	0	0	0.2	0
Beryllium	mg/l	0	0	0	0	0	0	0
Boron	mg/l	0	0	0	0	0.2	0	0.6
Cadmium	mg/l	0	0	0	0	0	0	0
Calcium	mg/l	110	50	60	70	83	90	70
Chromium	mg/l	0	0	0	0	0	0	0
Cobalt	mg/l	0	0	0	0	0	0	0
Copper	mg/l	0.6	0.5	0.5	1.3	0.52	1.8	0.4
Iron	mg/l	42	42	28	29	29	18	8
Lead	mg/l	0	0	0	0	0	0	0
Magnesium	mg/l	140	140	110	110	97	80	60
Manganase	mg/l	1.1	1.1	0.8	0.8	0.71	0.6	0.3
Molybdenum	mg/l	0	0	0	0	1	0	0
Nickel	mg/l	0	0	0	0.2	0.07	0	0
Phosphorus	mg/l	1830	2090	1870	1970	2000	1890	1890
Potassium	mg/l	3100	2500	2300	2500	2600	2100	2400
Silicon, soluable	mg/l	12	16	13	15	14	17	30
Silver	mg/l	0.2	0	0	0	0	0	0
Sodium	mg/l	1900	1600	1800	1800	1600	1500	1600
Strontium	mg/l	0	0.1	0.1	0	0.09	0.1	0
Sulfate	mg/l	4400	4700	3800	4400	4900	3400	3600
Sulfur	mg/l	1500	1600	1300	1500	1600	1100	1200
Titanium	mg/l	0.1	0	0	0	0	0	0
Vanadium	mg/l	0	0	0	0	0	0	0
Zinc	mg/l	10	7.3	2	4.1	2.7	3	0.9
Zirconium	mg/l	0.3	0	0	0.1	0.06	0	0
Eh	mv	149	469	71	161.2	518.3	444	495

Table 4.2 Swine (Pit No. 2) Leachate Chemistry

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	34200	36600	40000	40000	47000	47000	51400
Carbonate	mg/L	0	0	0	0	0	0	0
Chloride	mg/L	2855	2700	2570	2490	2360	2650	2140
Hydroxide	mg/L	0	0	0	0	0	0	0
pH	pH Units	6.47	6.57	6.77	6.82	6.72	6.86	6.66
Specific Conductivity	uS/cm	33200	35500	40400	41900	42800	46600	48700
Total Alkalinity	mg/L	28000	29200	32800	32800	38500	38500	42100
Ammonia as Nitrogen	mg/L	2400	5831	12600	10500	8490	15200	16300
Nitrate + Nitrite nitrogen	mg/L	25	6.6	7.2	3.6	3.8	4.7	0.83
Total Kjeldahl Nitrogen	mg/L	17100	19000	15100	19200	18800	17200	12000
Inorganic Carbon	mg/L	6730	7000	7670	7870	9250	9250	10100
Organic Carbon	mg/L	44000	48000	48000	54000	56000	68000	70000
Aluminum	mg/L	0.7	2.1	0.9	1	0.5	0.9	0
Barium	mg/L	0	0.8	0.5	0	0	0	0
Beryllium	mg/L	0	0	0	0	0	0	0
Boron	mg/L	0	0	0	0	0	0	1.6
Cadmium	mg/L	0	0	0	0	0	0	0
Calcium	mg/L	70	30	40	40	65	50	30
Chromium	mg/L	0	0.1	0	0.1	0	0	0
Cobalt	mg/L	0.1	0	0	0	0	0	0
Copper	mg/L	0.3	1	1.3	1.4	2.7	1.7	0.6
Iron	mg/L	70	37	54	35	29	19	10
Lead	mg/L	0	0	0	0	0	0	0
Magnesium	mg/L	80	30	30	20	20	20	10
Manganase	mg/L	2.6	0.5	0.4	0.3	0.12	0.1	0
Molybdenum	mg/L	0.4	0	0	0	0.57	0	0.2
Nickel	mg/L	0.3	0.2	0	0	0	0	0
Phosphorus	mg/L	1720	1380	1870	1520	1540	1540	1460
Potassium	mg/L	2800	2100	3200	2300	2600	2300	2400
Silicon, soluable	mg/L	32	13	15	12	11	12	48
Silver	mg/L	0	0.1	0	0	0	0	0
Sodium	mg/L	2300	1700	2600	2200	1700	1800	1700
Strontium	mg/L	0.2	0.1	0.2	0	0.08	0	0
Sulfate	mg/L	3500	3600	5700	5900	5800	2100	3800
Sulfur	mg/L	1200	1200	1900	2000	1900	690	1300
Titanium	mg/L	0	0.1	0	0	0	0	0
Vanadium	mg/L	0	0	0	0	0	0	0
Zinc	mg/L	2.6	2.5	2.9	3.3	2.1	1.8	1.4
Zirconium	mg/L	0	0.2	0	0	0	0	0
Eh	mv	-1.3	446	0	78.4	144	398	488

Table 4.3 Swine (Pit No. 3) layered with in situ soil leachate chemistry

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	33900	34600	36200	37100	40300	40600	45900
Carbonate	mg/L	0	0	0	0	0	0	0
Chloride	mg/L	3065	2830	2530	2660	2720	3300	2280
Hydroxide	mg/L	0	0	0	0	0	0	0
pH	pH Units	6.13	6.16	6.34	6.4	6.42	6.52	6.43
Specific Conductivity	uS/cm	29800	34600	36800	36400	36100	41800	41300
Total Alkalinity	mg/L	27800	28400	29700	30400	33000	33300	37600
Ammonia as Nitrogen	mg/L	4000	5395	10400	9290	7130	10000	12500
Nitrate + Nitrite nitrogen	mg/L	24	13	5.2	3.3	4.6	6.9	1.2
Total Kjeldahl Nitrogen	mg/L	19300	22000	19700	15400	20200	17100	8840
Inorganic Carbon	mg/L	6670	6310	7120	7300	7930	7990	9030
Organic Carbon	mg/L	68000	59000	56000	66000	59000	75000	64000
Aluminum	mg/L	0.8	1.4	0.7	0.8	0.5	1	0
Barium	mg/L	0	0.7	0.5	0.1	0	0	0
Beryllium	mg/L	0	0	0	0	0	0	0
Boron	mg/L	0.5	0	0	0	0	0	0
Cadmium	mg/L	0	0	0	0	0	0	0
Calcium	mg/L	20	100	90	80	100	90	70
Chromium	mg/L	0	0	0.2	0	0	0	0
Cobalt	mg/L	0	0.1	0.2	0	0.06	0	0
Copper	mg/L	0.7	1.3	1.6	1.9	1.5	1.6	1.1
Iron	mg/L	58	83	64	60	54	44	21
Lead	mg/L	0	0	0	0	0	0	0
Magnesium	mg/L	10	80	60	50	30	30	20
Manganase	mg/L	0.3	3.3	2.1	2	1.3	1.4	0.4
Molybdenum	mg/L	0.5	0	0	0	0.15	0	0
Nickel	mg/L	0	0.4	0.2	0.2	0.28	0.2	0.2
Phosphorus	mg/L	1170	2150	1860	1820	1800	1850	1580
Potassium	mg/L	2700	2600	2400	2500	2500	2200	2300
Silicon, soluable	mg/L	20	43	41	42	40	47	46
Silver	mg/L	0	0	0	0	0	0	0
Sodium	mg/L	2300	2200	2700	2200	1900	2100	2000
Strontium	mg/L	0.2	0.4	0.3	0.3	0.31	0.3	0.2
Sulfate	mg/L	5500	4600	3700	3900	4700	3100	3100
Sulfur	mg/L	1800	1500	1200	1300	1600	1000	1000
Titanium	mg/L	0	0.1	0	0	0.07	0	0
Vanadium	mg/L	0	0	0	0	0	0.1	0
Zinc	mg/L	3.9	3.6	3.2	3.1	2.4	2.2	1.5
Zirconium	mg/L	0	0	0	0	0	0	0
Eh	mv	39.2	486	59	111.3	123.9	361	504

Table 4.4 Bovine (Pit No. 4) Leachate Chemistry

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	35100	40700	43100	45600	53400	48700	50100
Carbonate	mg/L	0	0	0	0	0	0	0
Chloride	mg/L	2605	2765	2400	2310	2450	3810	2180
Hydroxide	mg/L	0	0	0	0	0	0	0
pH	pH Units	6.83	6.81	7.9	7.01	6.98	6.89	6.86
Specific Conductivity	uS/cm	32700	32700	41500	42100	46900	50000	51100
Total Alkalinity	mg/L	28800	33400	35300	37400	43800	39900	41100
Ammonia as Nitrogen	mg/L	5200	4708	13200	13900	10200	13000	19200
Nitrate + Nitrite nitrogen	mg/L	23	14	2.1	2.6	6.7	3.5	1.1
Total Kjeldahl Nitrogen	mg/L	18300	11000	16000	19400	20100	17600	18400
Inorganic Carbon	mg/L	6900	8010	8480	8970	10400	9580	9860
Organic Carbon	mg/L	43000	46000	47000	57000	60000	68000	75000
Aluminum	mg/L	1.7	1.2	0	0.8	0.4	1	0
Barium	mg/L	0.3	0.6	0.5	0.2	0.06	0	0
Beryllium	mg/L	0	0	0	0	0	0	0
Boron	mg/L	0	0.7	0.9	0.7	0.6	1.2	0.7
Cadmium	mg/L	0	0	0	0	0	0	0
Calcium	mg/L	60	30	20	40	38	50	20
Chromium	mg/L	0	0	0	0	0	0	0
Cobalt	mg/L	0.1	0	0	0	0	0	0
Copper	mg/L	0.6	1.1	1	1	0.6	0.7	0.5
Iron	mg/L	110	43	46	29	28	15	9.8
Lead	mg/L	0	0	0	0	0	0	0.9
Magnesium	mg/L	30	10	30	20	15	30	10
Manganase	mg/L	0.5	0.3	0.4	0.4	0.19	0	0.1
Molybdenum	mg/L	1.8	0	0.2	0	0.21	0	0
Nickel	mg/L	0.4	0.1	0	0	0.1	0	0
Phosphorus	mg/L	920	1300	1300	1230	1170	1200	1080
Potassium	mg/L	1900	2100	2100	2100	2200	1800	1900
Silicon, soluable	mg/L	29	25	27	26	22	27	29
Silver	mg/L	0	0	0	0	0	0	0
Sodium	mg/L	1600	2000	2700	2200	1900	2000	2000
Strontium	mg/L	0.7	0.3	0.3	0.3	0.26	0.3	0.1
Sulfate	mg/L	3700	4500	6300	6800	4600	1700	2400
Sulfur	mg/L	1200	1500	2100	2300	1500	580	810
Titanium	mg/L	0.2	0	0	0	0	0	0
Vanadium	mg/L	0	0	0	0	0	0	0
Zinc	mg/L	3.5	4.2	3.3	2.9	2.3	1.8	1.1
Zirconium	mg/L	0.2	0	0	0	0	0	0
Eh	mv	25.7	451	31	89	72.1	343	401

Table 4.5 Bovine (Pit No. 5) layered with in situ soil leachate chemistry

DATE of SAMPLE		Aug. 17 2005	Sept. 21 2005	Oct. 20 2005	Nov. 23 2005	May 25 2006	Oct. 26 2006	Sept. 11 2007
Bicarbonate	mg/L	20900	38600	40500	41000	53400	56500	58700
Carbonate	mg/L	0	0	0	0	0	0	0
Chloride	mg/L	1775	2200	1830	1630	1940	1870	1800
Hydroxide	mg/L	0	0	0	0	0	0	0
pH	pH Units	6.49	6.92	7.07	7.28	6.95	7.05	6.92
Specific Conductivity	uS/cm	22300	31200	44000	42600	46000	49300	55400
Total Alkalinity	mg/L	17100	31600	33200	33600	43800	46300	48100
Ammonia as Nitrogen	mg/L	1400	3770	10800	11900	8100	20900	19200
Nitrate + Nitrite nitrogen	mg/L	22	33	6.9	2.4	4.9	4.5	0.46
Total Kjeldahl Nitrogen	mg/L	17200	13000	14500	16800	19000	20000	19100
Inorganic Carbon	mg/L	4110	7590	7970	8060	10500	11100	11500
Organic Carbon	mg/L	33000	48000	40000	44000	60000	71000	81000
Aluminum	mg/L	1.2	3.5	1.1	0.7	0	0.7	0
Barium	mg/L	0	1.1	0.7	0.5	0.07	0.3	0
Beryllium	mg/L	0	0	0	0	0	0	0
Boron	mg/L	0	0	0	0	0	0	0.4
Cadmium	mg/L	0	0	0	0	0	0	0
Calcium	mg/L	20	60	40	50	64	70	40
Chromium	mg/L	0	0	0	0.1	0	0	0.1
Cobalt	mg/L	0	0	0	0.1	0	0	0
Copper	mg/L	0.3	0.7	0.6	0.5	0.15	0.4	0.4
Iron	mg/L	46	90	50	32	10	806	5.6
Lead	mg/L	0	0	0	0	0	0	0
Magnesium	mg/L	30	10	10	10	12	10	0
Manganase	mg/L	0.3	0.5	0.3	0.4	0.24	0.3	0
Molybdenum	mg/L	1.3	0	0	0	0.1	0	0
Nickel	mg/L	0	0.3	0	0	0.12	0	0.1
Phosphorus	mg/L	1350	1290	1030	1120	1130	1250	990
Potassium	mg/L	2800	2300	2100	2300	2300	2100	2400
Silicon, soluable	mg/L	10	48	38	40	31	35	54
Silver	mg/L	0	0	0	0	0	0	0
Sodium	mg/L	2100	2000	2100	2100	1700	2000	2000
Strontium	mg/L	0	0.7	0.7	0.8	0.5	0.5	0.3
Sulfate	mg/L	4600	4500	5800	8800	5300	1700	2200
Sulfur	mg/L	1500	1500	1900	2900	1800	560	740
Titanium	mg/L	0	0.3	0.1	0.1	0	0	0
Vanadium	mg/L	0	0	0	0	0	0	0
Zinc	mg/L	2.3	4.6	2.9	13	1.9	1.7	0.8
Zirconium	mg/L	0	0	0	0	0	0	0
Eh	mv	1	474	0	52.1	49.1	241	498

4.1.1 Ammonia-N

As the livestock carcass decomposed, ammonia levels begin to rise. Figure 4.1 shows the trend over the two year sampling period for levels of ammonia. In the early stages of decomposition, up to the first two months, ammonia levels were at their lowest around 5,000 mg/l for each species. Ammonia continued in an average upward trend for all sample dates. At two years, bovine had the highest concentration at 19,200 mg/l with swine following closely at 16,300 mg/l and poultry having the lowest ammonia concentration at 10,100 mg/l. The chemistry results from SRC Analytical Laboratory are accurate to plus/minus 100 mg/l.

Since Nitrogen is the second most abundant element in the body, the trend of Figure 4.1 approximately shows the breakdown of different cellular systems in the livestock. The first period of sampling between 1-2 months depicts the excretion of components in the digestive system such as urine and manure. The second period between 2-4 months depicts the breakdown of blood and liquids in the animal. Between 4-14 months, it becomes harder to distinguish what is actually happening in the decomposition process. The breakdown of heavy proteins such as muscles and bone marrow is a possible route of heavy nitrogen appearing in the analysis at those time periods (Vass 2001).

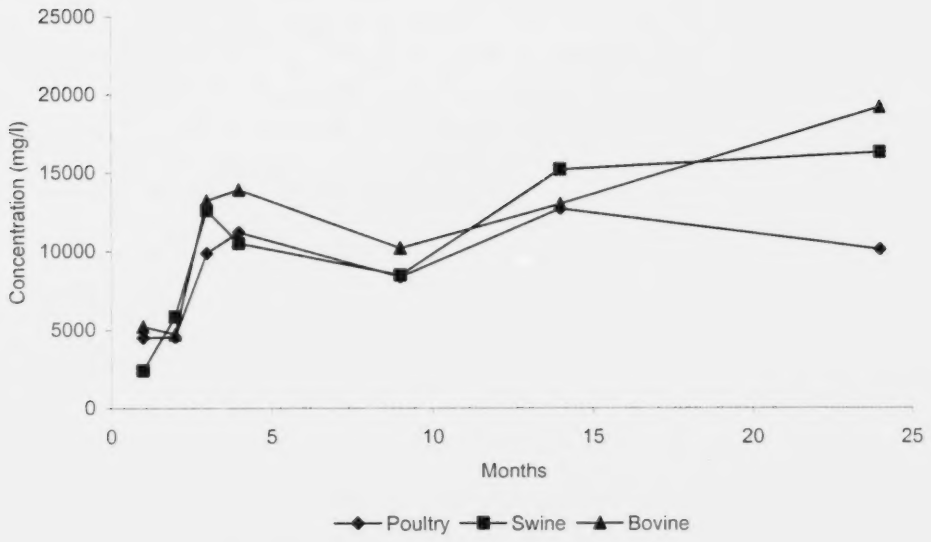

Figure 4.1 Ammonia-N concentrations pure leachate

The results between the swine and bovine pits including soil with the carcasses follow the same general trend line. Figures 4.2 and 4.3 are an illustration of those results. The swine layered with soil have slightly lower concentration of ammonia at all sample dates. At fourteen months there is approximately a 5,000 mg/l difference between the concentration of pure swine leachate and that which is layered in soil (Fig. 4.2). For bovine, the trend is not the same as swine, the bovine layered with soil is lower at only the first nine months of sampling. After nine months, the soil layered bovine show a concentration spike at fourteen months with a value of 20,900 mg/l. Overall, the average ammonia concentrations for all species after two years of sampling are 10,095 mg/l.

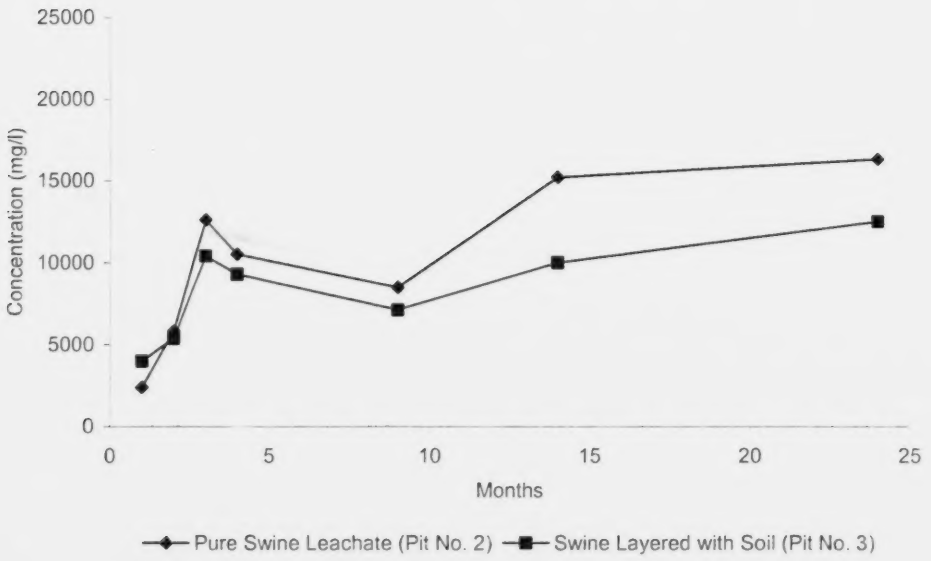

Figure 4.2 Ammonia-N concentrations for swine - pure leachate vs. soil layering

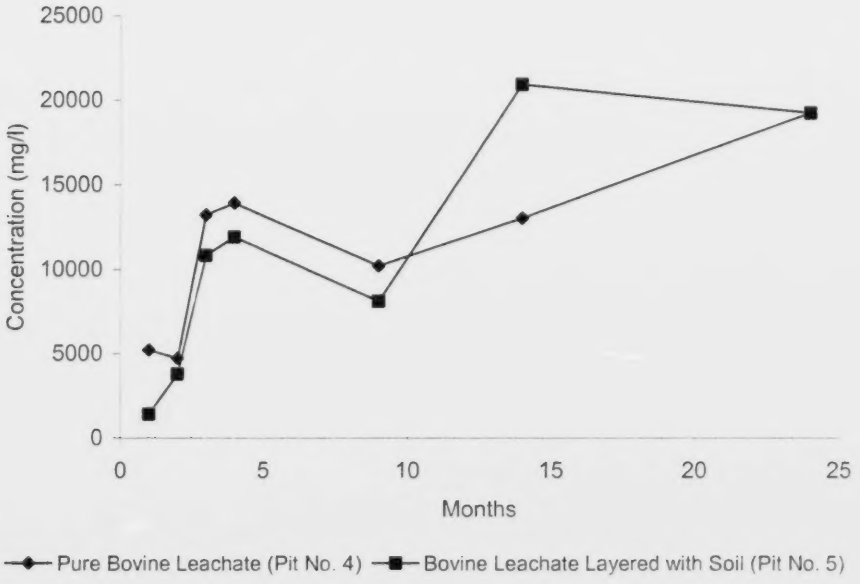

Figure 4.3 Ammonia-N concentrations for bovine - pure leachate vs. soil layering

4.1.2 Bicarbonate

Bicarbonate (HCO_3^-) values during the sampling period of each species of livestock followed a similar trend line (Fig. 4.4). Poultry (Pit No. 1) started with a concentration of 25,700 mg/l and after two years had a concentration of 39,600 mg/l. Swine (Pit No. 2) had a concentration of 34,200 mg/l at the first sample date and ended with a concentration of 51,400 mg/l after two years of decomposition. Bovine (Pit No. 4) started at 35,100 mg/l and rose to a peak concentration at nine months at 53,400 mg/l and decreased to 50,100 mg/l after 24 months. If all species are averaged over the sample period, an estimated amount of bicarbonate for any type of livestock, may be approximately 40,600 mg/l.

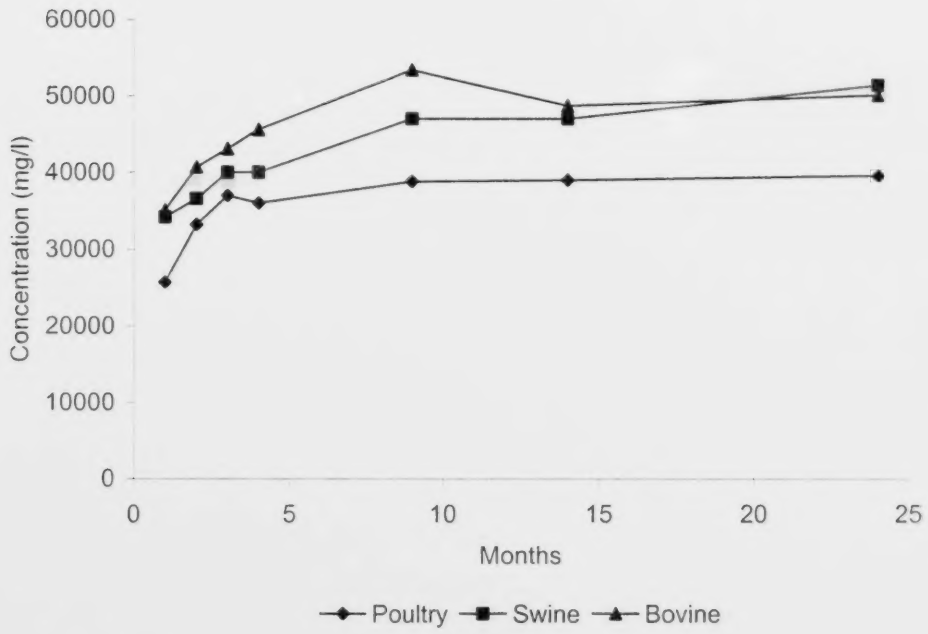

Figure 4.4 Bicarbonate concentrations pure leachate

The results for swine and bovine pits layered with soil are shown in figures 4.5 and 4.6. At the first sample date, each swine pit had similar concentration values, from the first sample date on, pure swine leachate had a slightly higher concentration than that of the swine layered in soil. By the end of the sampling period, the difference between the two concentrations was 5,500 mg/l. In the case of the bovine pits, the pure bovine leachate had a higher value on the first sample date than that of the bovine layered in soil. The difference got smaller by the second sample date, and by the ninth month, each pit had the same bicarbonate concentration. From nine months on, the bovine layered in soil had a slightly higher concentration of bicarbonate. The difference at 24 months between the pure bovine leachate and that layered in soil was 8,600 mg/l.

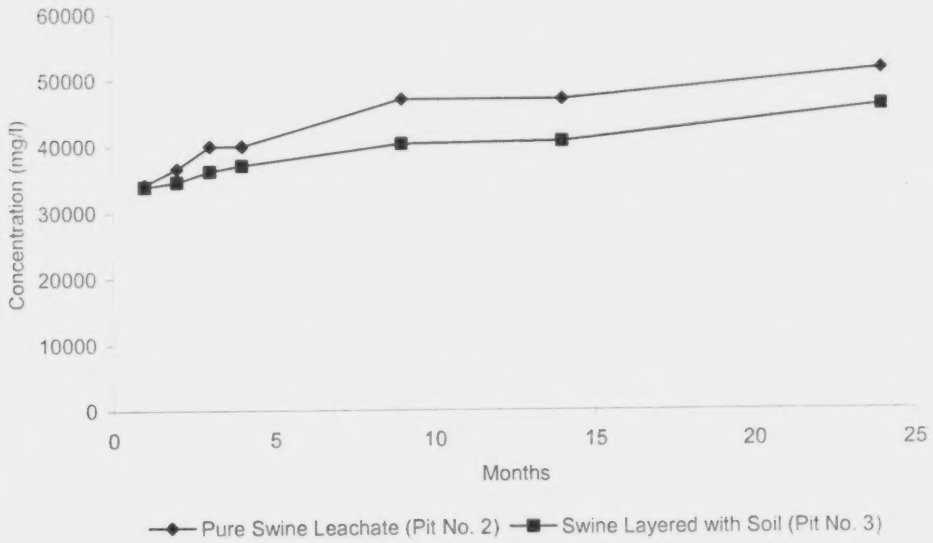

Figure 4.5 Bicarbonate concentrations for swine - pure leachate vs. soil layering

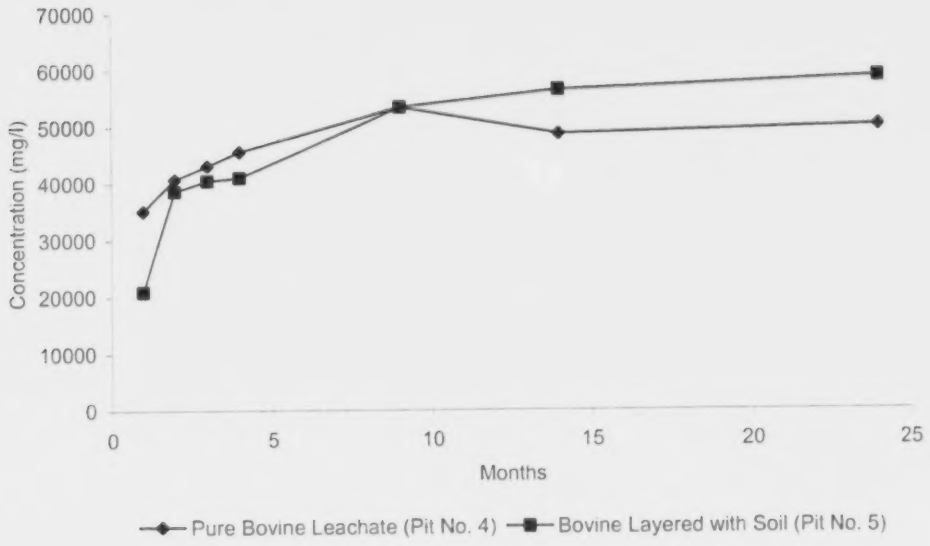

Figure 4.6 Bicarbonate concentrations for bovine - pure leachate vs. soil layering

4.1.3 Sulphate

Sulphate (SO_4^{-2}) for each species follows the same general trend for the sample period (Fig. 4.7). Poultry at the first sample date started with a concentration of 4,400 mg/l and rose slightly by the ninth month. After nine months, the poultry leachate showed concentrations less than the initial values. By 24 months, the sulphate concentration for poultry was 3,600 mg/l. Swine initially had a concentration of 3,500 mg/l and rose sharply the first four months to 5,900 mg/l. After four months, the levels of sulphate in the swine leachate decreased to 2,100 mg/l at fourteen months and rose slightly to 3,800 mg/l at 24 months. Bovine reached the highest value at 6800 mg/l within four months of burial and after 24 months had a concentration of 2,400 mg/l. The chemistry results for sulphate are accurate to within 20 mg/l.

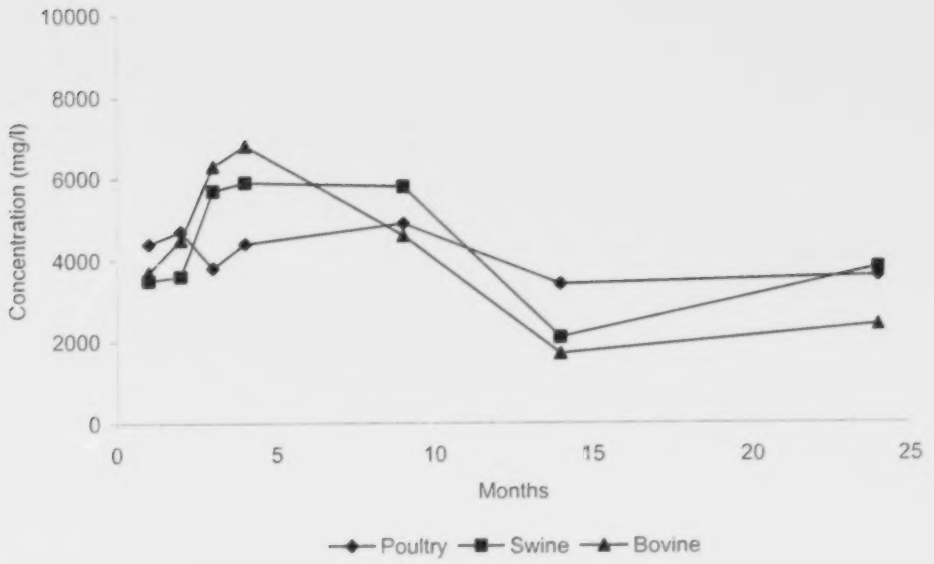

Figure 4.7 Sulphate concentrations pure leachate

For Pit No. 3 & 5 consisting of swine and bovine layered with soil, shows slight differences in leachate concentrations. Swine leachate concentrations shown in Figure 4.8 have an initial separation of 2,000 mg/l. After the first sample date, pure swine leachate increased in concentration while the swine layered in soil decreased in sulphate concentration. Throughout the sample period, the pure swine leachate crossed the trend line of leachate in layered soil three times. By 24 months, pure swine leachate had a higher concentration value at 3,800 mg/l compared to 3,100 mg/l. The two pits of bovine follow each other's trend lines more closely than that of the swine (Fig. 4.9). Initially the bovine layered in soil had a higher concentration than pure bovine leachate. After four months, this was still the case with a peak concentration of 8,800 mg/l reached by the bovine in layered soil. After 24 months, concentrations in each had decreased to nearly the same value of 2,400 mg/l.

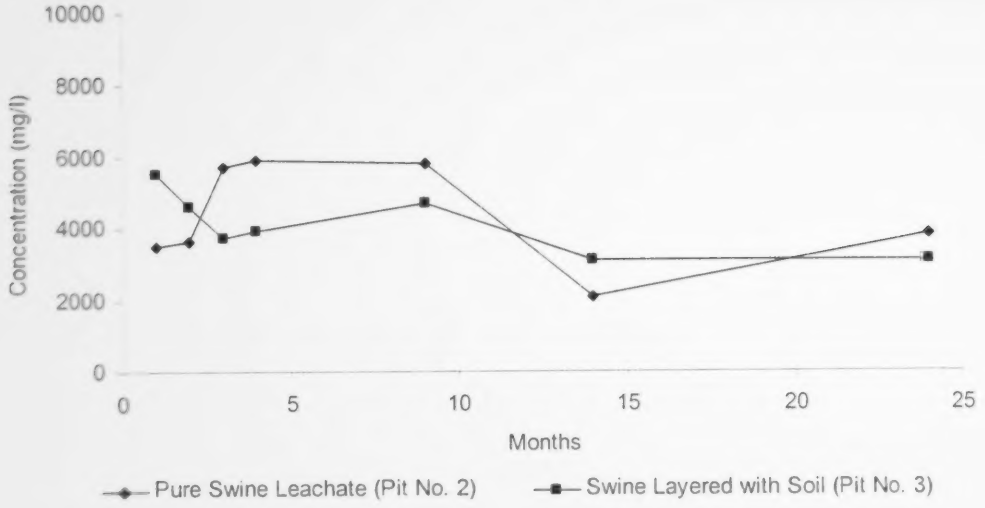

Figure 4.8 Sulphate concentrations for swine – pure leachate vs. soil layering

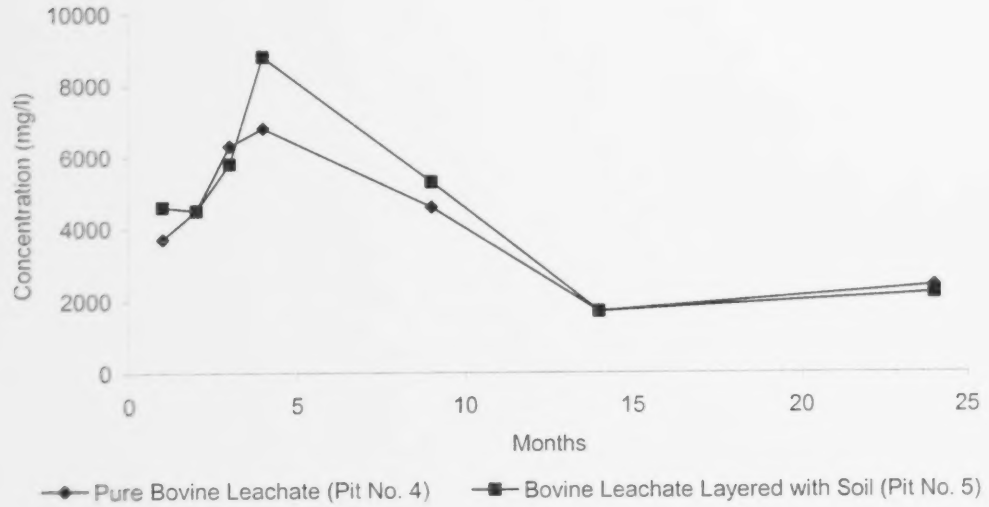

Figure 4.9 Sulphate concentrations for bovine – pure leachate vs. soil layering

4.1.4 Sodium

Sodium is another constituent of in livestock leachate. Each species analyzed presents

similar concentrations of sodium for all sample points and chemistry results for sodium are accurate to within 100 mg/l. Bovine initially had the lowest concentration of sodium and increased to 2,700 mg/l by the third month. After four months, sodium concentrations in bovine leachate decreased to 2,200 mg/l and remained nearly constant for the remainder of the sampling period. Swine initially had the highest concentration of sodium at the first sample date at 2,300 mg/l. By the fourth month, the concentrations in the swine leachate slightly increased and then followed the same trend as bovine with a concentration around 2,200. Poultry began with a concentration of 1,900 mg/l, and slightly decreased to approximately 1,600 mg/l for the remainder of the sampling period.

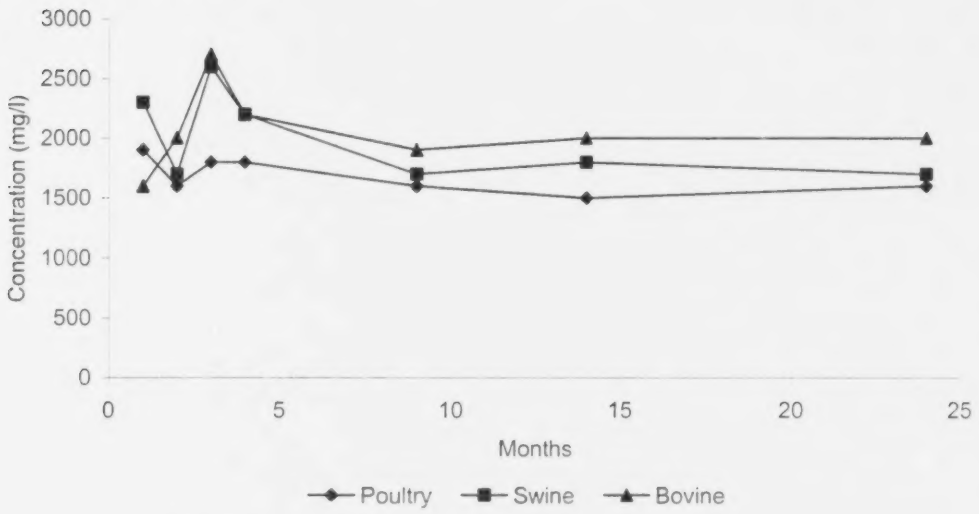

Figure 4.10 Sodium concentrations pure leachate

For Pit No. 3 & 5 consisting of swine and bovine layered with soil, shows slight differences in leachate sodium concentrations. Sodium concentrations in pure swine leachate and that of swine layered in soil follow the same general trend line over the sample period. Swine layered in soil showed slightly higher concentrations of sodium over the sample period and after 24 months, showed a difference of only 300 mg/l. In

the case of pure bovine leachate and bovine layered in soil, they also follow each other quite closely. Initially pure bovine leachate peaked at 2,700 mg/l after three months and then levelled off to 2,000 mg/l by fourteen months. At fourteen months, both sample sets had equal values.

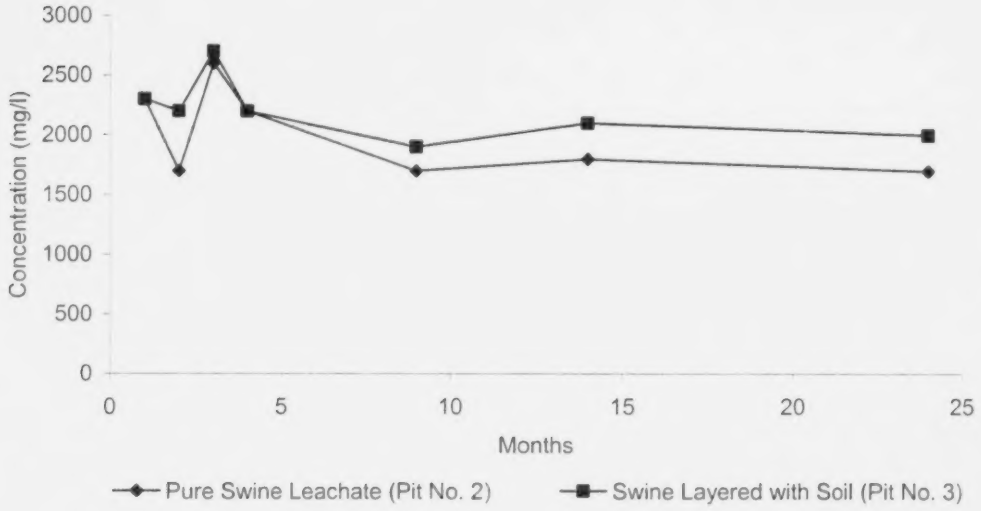

Figure 4.11 Sodium concentrations for swine – pure leachate vs. soil layering

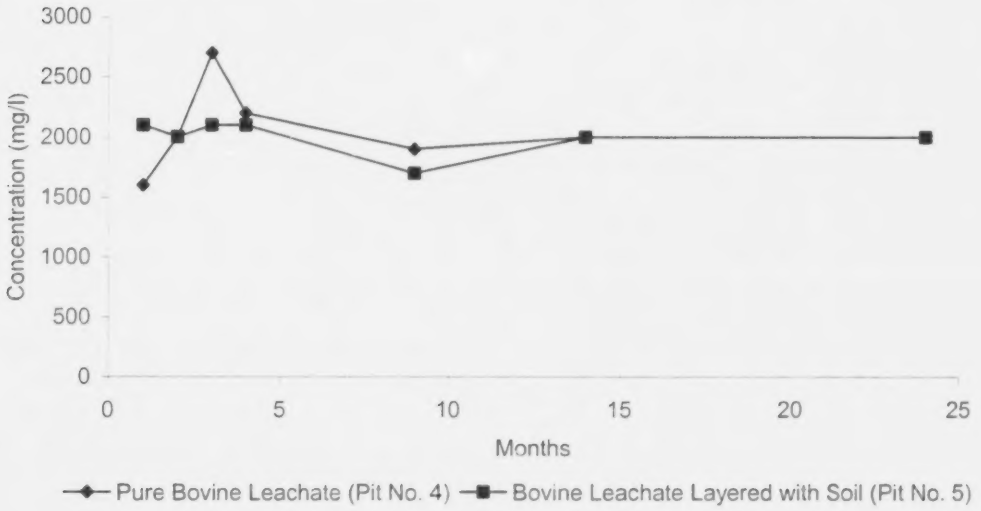

Figure 4.12 Sodium concentrations for bovine – pure leachate vs. soil layering

4.1.5 Chloride

Chloride concentrations for each species generally follow the same trend and have similar concentrations for all sample dates. Bovine reached the highest concentration at fourteen months with a chloride concentration of 3810 mg/l. An average chloride value for livestock decomposition leachate was calculated as 2704 mg/l (Fig. 4.13).

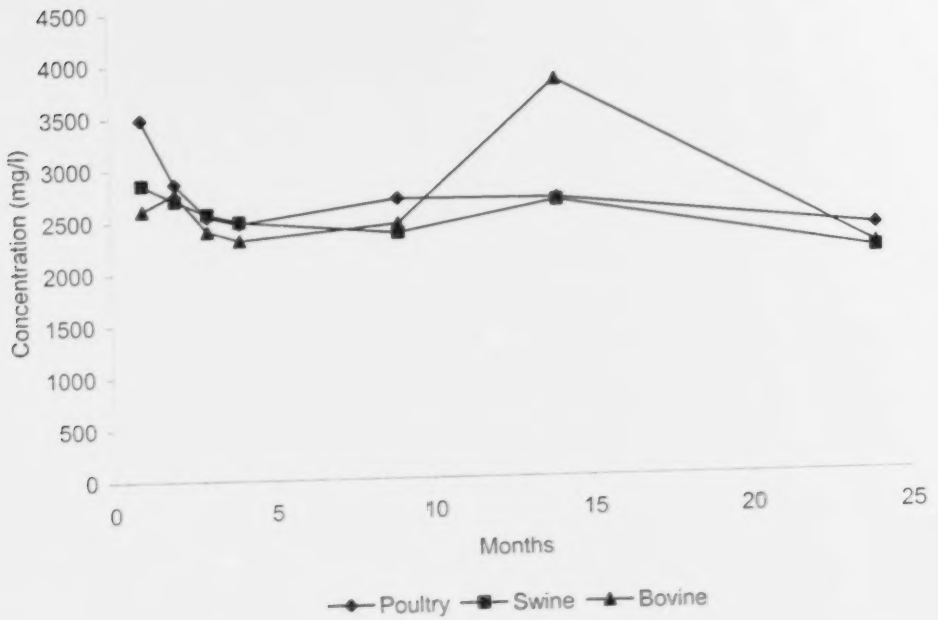

Figure 4.13 Chloride concentrations pure leachate

Chloride concentrations for pure swine and swine layered in soil (Fig. 4.14), follow the same general trend and their initial difference in concentration was 200 mg/l while after 24 months they only differed by 140 mg/l. For the case of pure bovine leachate and bovine layered in soil (Fig. 4.15), the initial separation in concentrations of chloride was nearly 900 mg/l. After 24 months, they differed by 380mg/l with pure bovine leachate

having a higher concentration than that of bovine layered in soil for the entire sampling period.

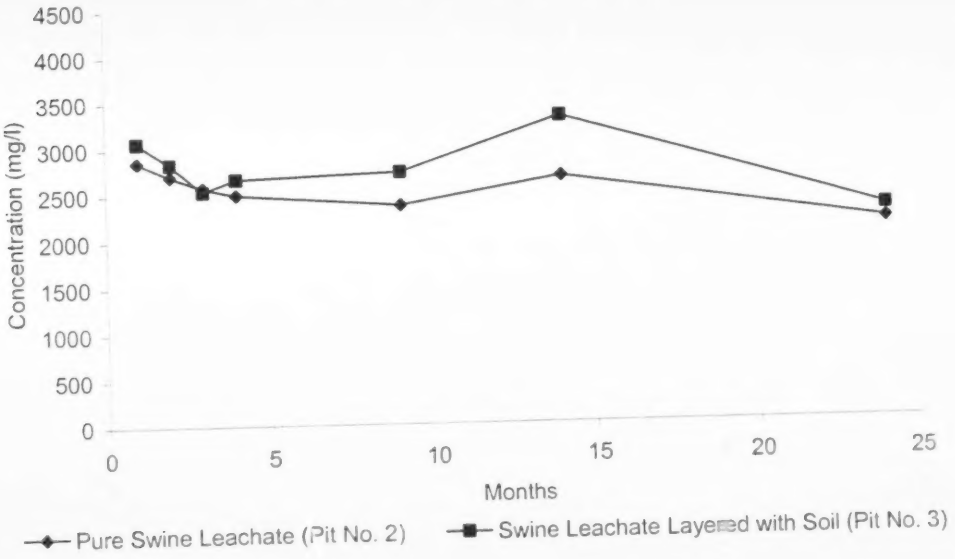

Figure 4.14 Chloride Concentrations for swine – pure leachate vs. soil layering

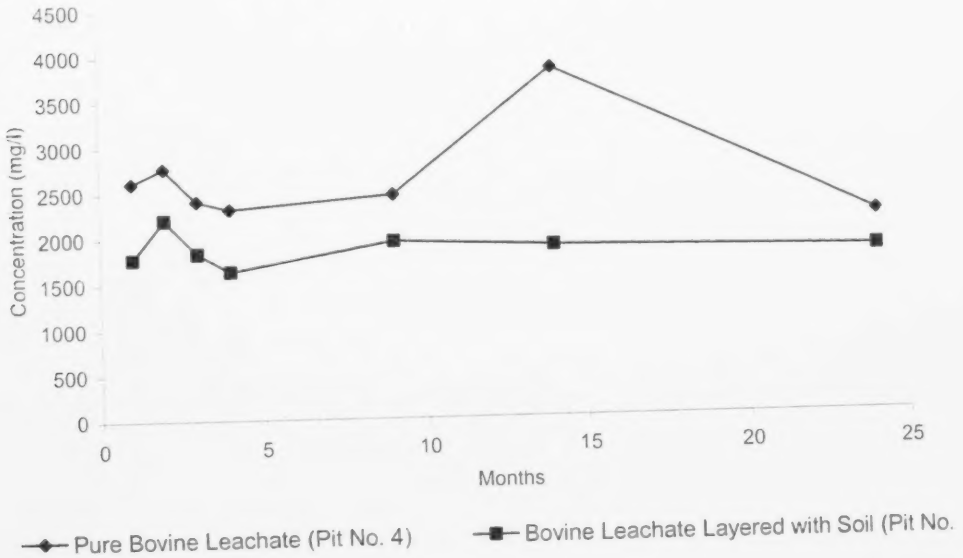

Figure 4.15 Chloride Concentrations for bovine – pure leachate vs. soil layering

4.1.6 Potassium

Potassium concentrations for each species do not fluctuate much during the 24 month sample period (Fig. 4.16). During the entire sample period, bovine had the lowest average concentration and poultry and swine had a slightly higher concentration. The breakdown of proteins is a major source of potassium production in the leachate (Vass 2001). The average potassium value for the three species during the sample period is 2370 mg/l with little effect caused by soil layered in the swine and bovine pits (Fig. 4.17 and 4.18).

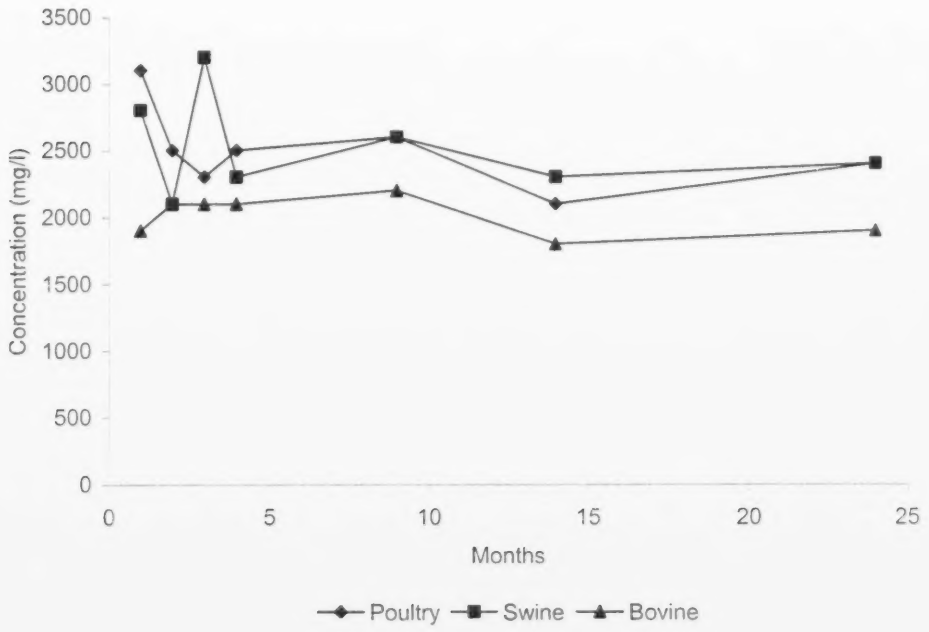

Figure 4.16 Potassium concentrations pure leachate

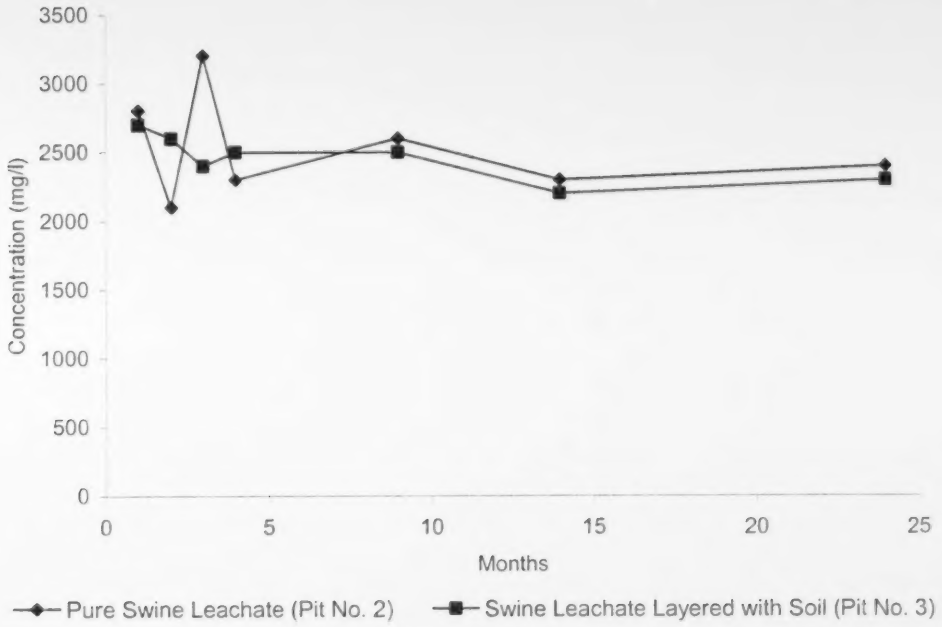

Figure 4.17 Potassium concentrations for swine – pure leachate vs. soil layering

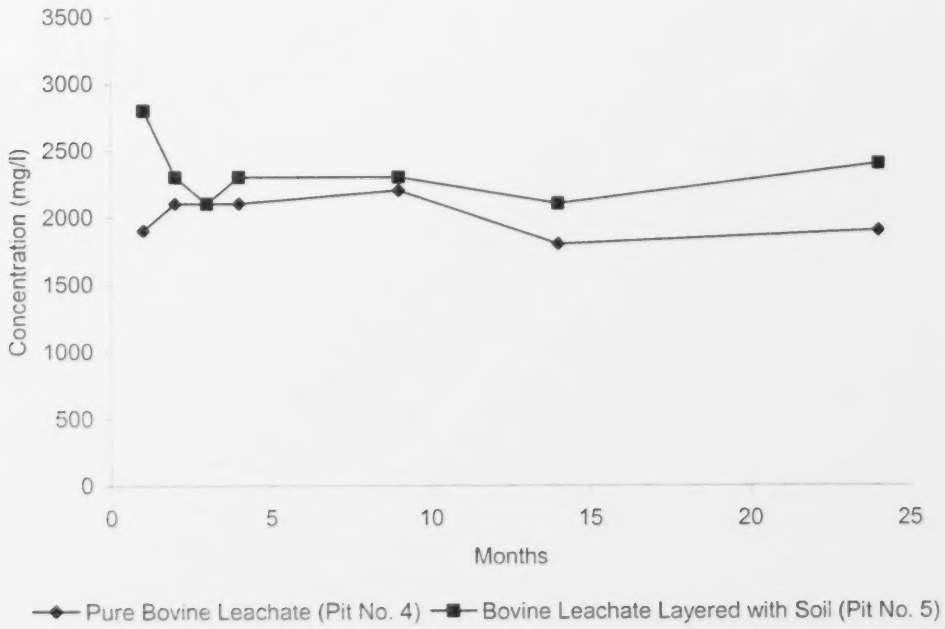

Figure 4.18 Potassium concentrations for bovine – pure leachate vs. soil layering

4.1.7 Phosphorous

Phosphorous concentrations for each species generally follow the same trend of slight fluctuations in early sample dates to a levelled value by fourteen months. Poultry contain the highest amount of phosphorous with a maximum value of 2,090 mg/l. Swine had a maximum value of 1,870 mg/l and bovine the lowest at a maximum value of 1,300 mg/l. Differences in the values could possibly be explained by diet and biological attributes of the species. Poultry, in general have significantly smaller frame and due to the large amount of feathers have more protein structures that include phosphorous molecules. As for differences between pure swine leachate and swine layered in soil, the values are within 400 mg/l for each sample date and follow the same trend. In the case of bovine, the concentrations are within 200 mg/l for nearly all sample dates and follow the same basic trend.

To determine how the phosphorus is present in the leachate solution, the mortality chemistry data was entered into PHREEQC (Parkhurst & Appelo 1999) and speciated. These results can be seen in Table 4.6. Phosphorus is mainly present in the leachate as phosphoric acid $H_2PO_4^-$, which is consistent with Moore et al. (2002) ionic presence of phosphorus in the body.

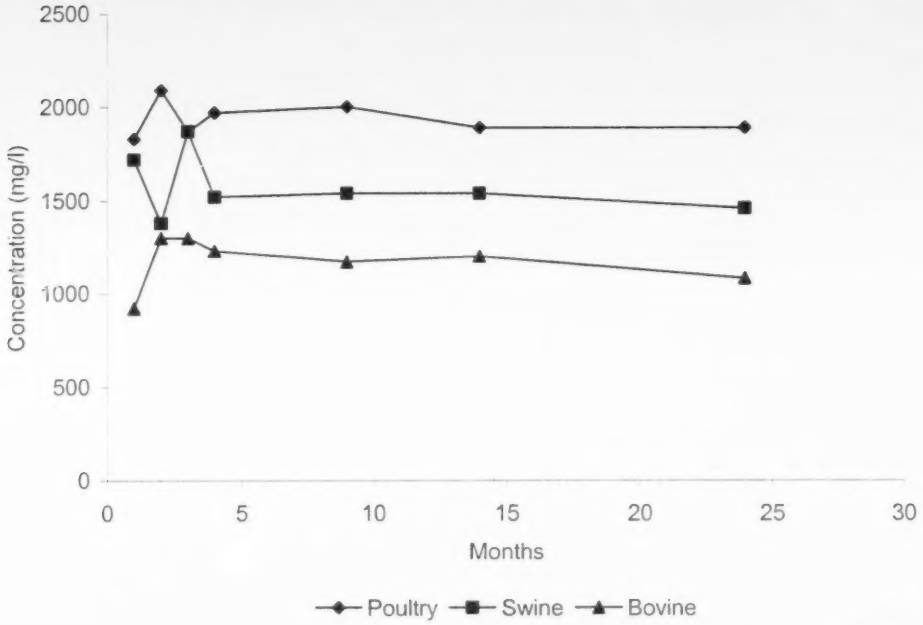

Figure 4.19 Phosphorus concentrations pure leachate

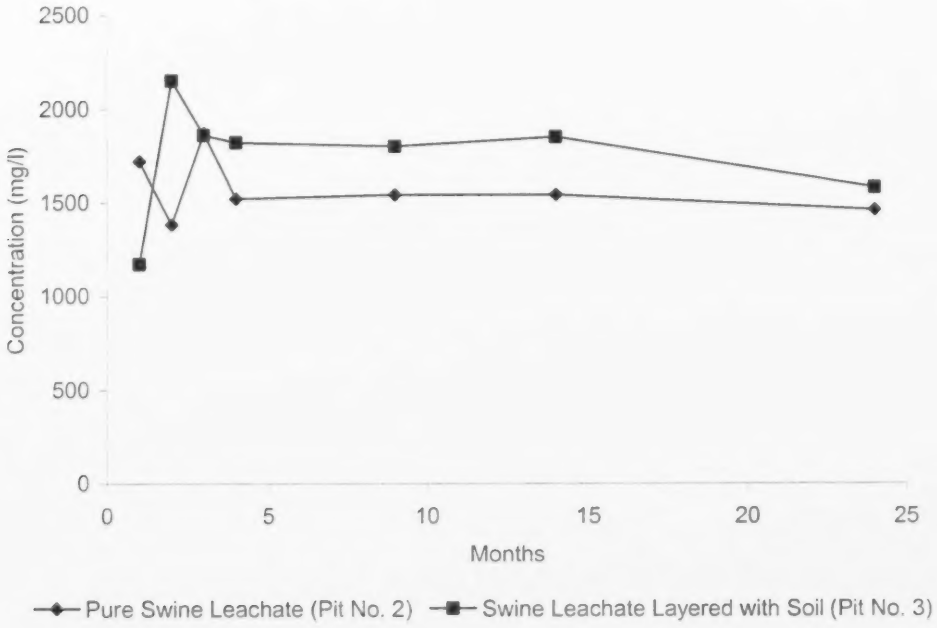

Figure 4.20 Phosphorus concentrations for swine – pure leachate vs. soil layering

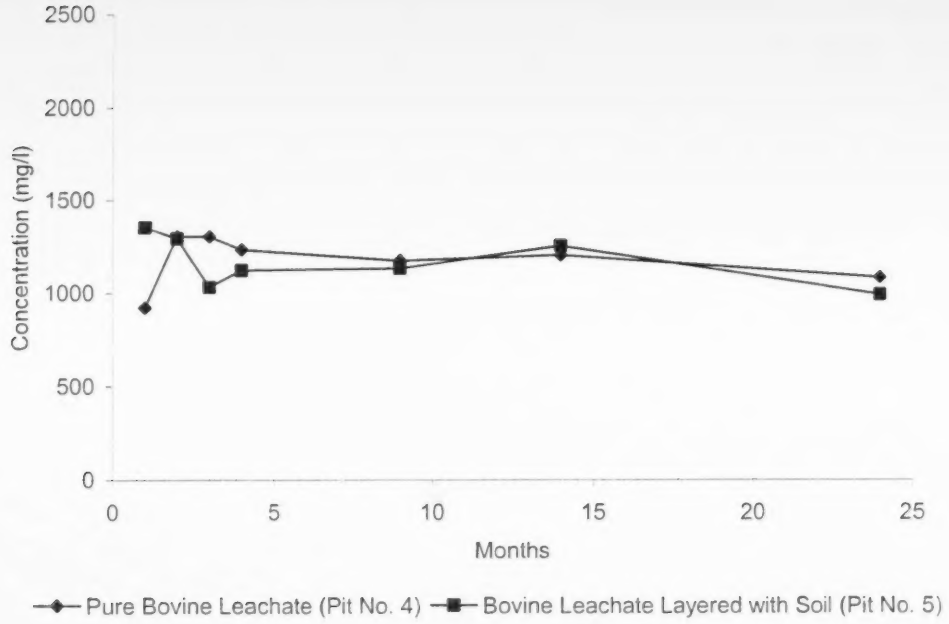

Figure 4.21 Phosphorus concentrations for bovine – pure leachate vs. soil layering

4.1.8 Chemistry Significance

Figure 4.22 and 4.23 show the difference between the results of this study to other potential contaminants such as earthen manure storages, landfill leachate, and the Scotland burial site study whose concentration values were taken from Tables 2.1 and 2.4. The results show; ammonium is of the most concern and is significantly higher in livestock leachate than hog manure and landfill leachate. The same is true for potassium, sodium, alkalinity, chloride, sulphate and phosphorus. Ammonia is important due to nitrogen's ability to convert from nitrogen to nitrate/nitrite. All values are much higher than recommended water quality standards seen in Table 2.3. Figure 4.23 is plotted with concentrations in meq/l to show how close the charge balance is between the anions and cations in the leachate solution.

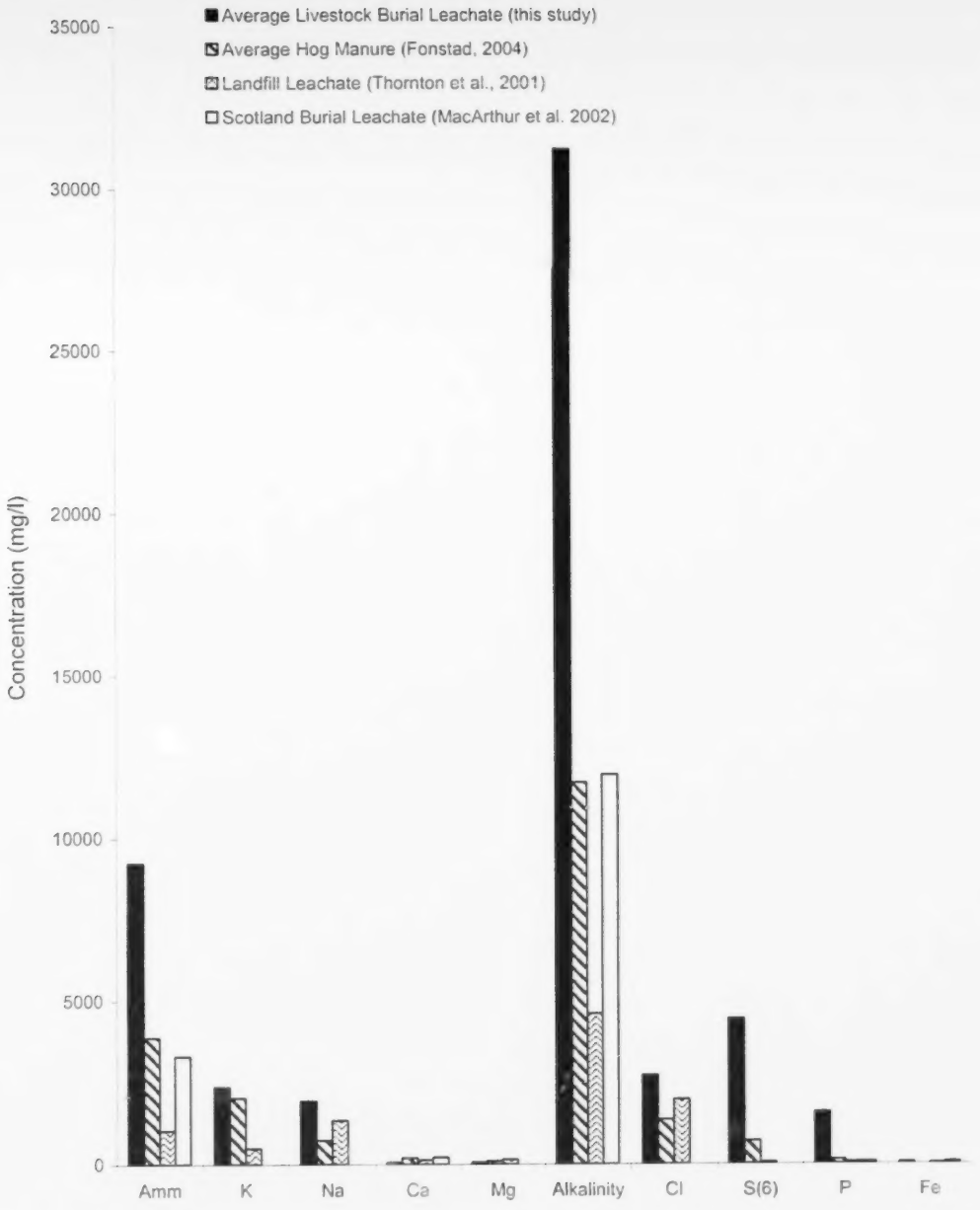

Figure 4.22 Leachate concentration comparisons (mg/l)

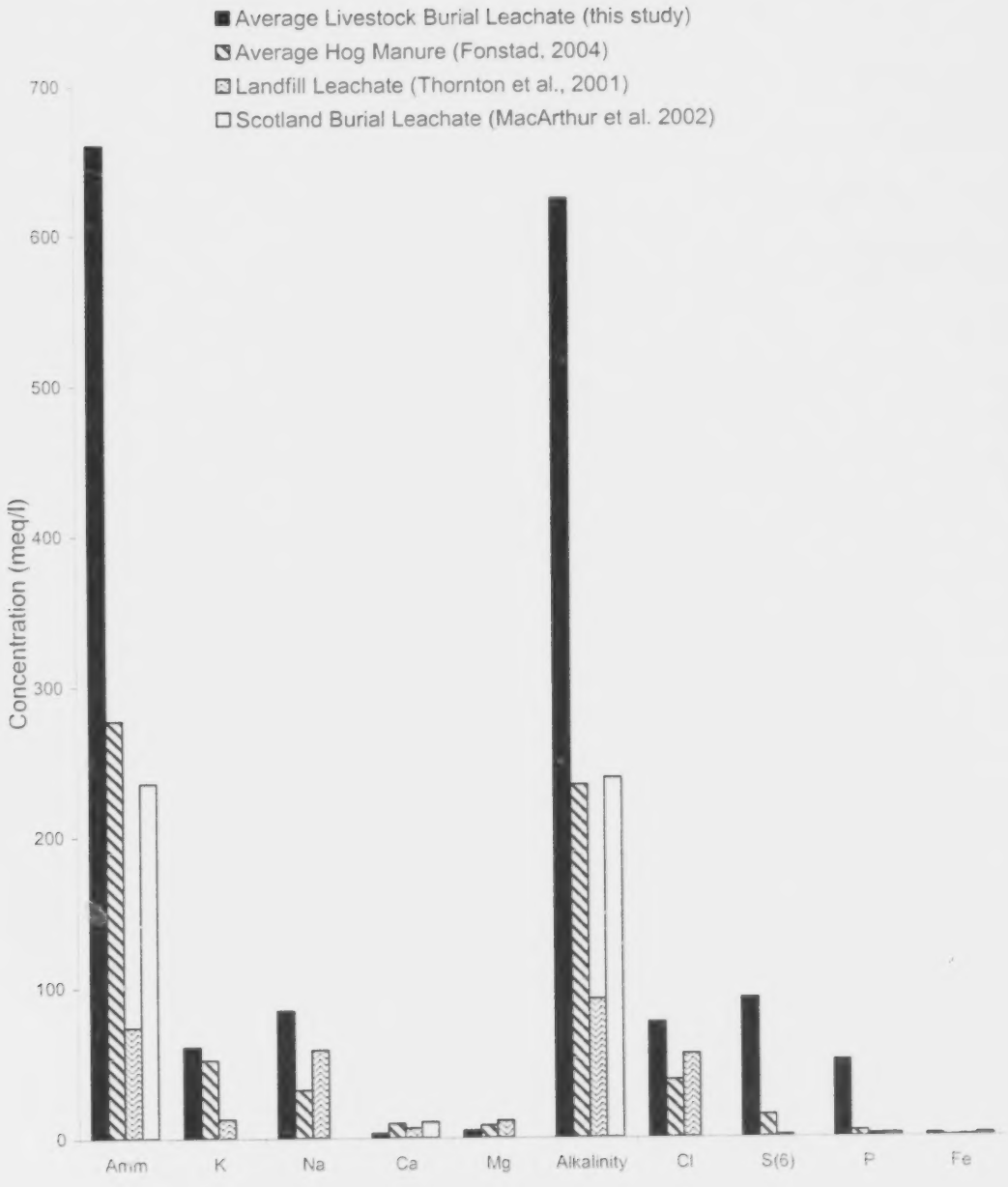

Figure 4.23 Leachate concentration comparisons (meq/l)

4.1.9 Leachate Chemistry Speciation

Speciation of the mortality leachate chemistry produced the following results shown in table 4.6. The results for phosphorus are consistent with the literature from Moore et al. (2002) showing that phosphorus is mainly present in the body in the form of phosphoric acids. Sulphate is speciated and found mostly in the form of ammonium sulphate, while Nitrogen is found commonly in the leachate as $NH4^{+}$. The calculated percent error by PHREEQC on this speciation is less than one percent.

Table 4.6 PHREEQC Speciation of Averaged Livestock Mortality Leachate Chemistry Concentrations

Species	Molality	Activity	Species	Molality	Activity
H+	2.67E-07	2.00E-07	MgSO4	1.20E-04	1.48E-04
OH-	8.29E-08	4.86E-08	H(0)	4.57E-09	
H2O	5.55E+01	9.69E-01	N(-3)	7.28E-01	
C(4)	7.93E-01		NH4+	7.04E-01	3.76E-01
HCO3-	6.25E-01	4.13E-01	NH4SO4-	2.27E-02	1.76E-02
CO2	1.55E-01	1.91E-01	NH3	8.72E-04	1.08E-03
NaHCO3	1.02E-02	1.25E-02	Na	8.79E-02	
MgHCO3+	1.08E-03	8.38E-04	Na+	7.58E-02	5.39E-02
CaHCO3+	7.67E-04	5.06E-04	NaHCO3	1.02E-02	1.25E-02
FeHCO3+	4.39E-04	3.41E-04	NaSO4-	1.27E-03	9.84E-04
FeHCO3+	4.39E-04	3.41E-04	NaHPO4-	5.70E-04	4.43E-04
Ca	1.45E-03		P	5.41E-02	
CaHCO3+	7.67E-04	5.06E-04	HPO4-2	3.07E-02	4.21E-03
CaHPO4	1.80E-04	2.22E-04	H2PO4-	2.15E-02	1.35E-02
Cl	7.93E-02		NaHPO4-	5.70E-04	4.43E-04
Cl-	7.93E-02	4.80E-02	MgHPO4	4.39E-04	5.42E-04
Fe(2)	6.84E-04		KHPO4-	3.99E-04	3.10E-04
FeHCO3+	4.39E-04	3.41E-04	CaHPO4	1.80E-04	2.22E-04
FeHPO4	1.12E-04	1.39E-04	FeHPO4	1.12E-04	1.39E-04
H(0)	4.57E-09		S(6)	4.80E-02	
K	6.41E-02		NH4SO4-	2.27E-02	1.76E-02
K+	6.25E-02	3.78E-02	SO4-2	2.26E-02	3.64E-03
KSO4-	1.24E-03	9.66E-04	NaSO4-	1.27E-03	9.84E-04
KHPO4-	3.99E-04	3.10E-04	KSO4-	1.24E-03	9.66E-04
Mg	2.35E-03		MgSO4	1.20E-04	1.48E-04
MgHCO3+	1.08E-03	8.38E-04	Si	3.70E-04	
Mg+2	5.91E-04	1.74E-04	H4SiO4	3.69E-04	4.55E-04
MgHPO4	4.39E-04	5.42E-04			

4.2 MIGRATE Model Results

This section presents the results found from the MIGRATE modeling software. Concentration profiles are presented as a ratio of computed concentration versus initial concentration (C/Co). The initial concentration was assumed to be an average ammonia value of 15000 mg/l with no attenuation included in the model. Boundary conditions were specified in section 2.4.1.3.

Figures 4.24 – 4.26 represent the concentration profiles for unattenuated ammonia transport for three ranges of flux values with a porosity of 0.3. For the low flux case simulating a clay soil (Fig. 4.24), it is apparent that the profile is diffusion dominated system, while Figures 4.25 and 4.26 are more typical profiles of an advection dominated system (till and sand) with plume differences between diffusion values. For the sand situation (Fig. 4.26), only one contour appears on the plot due to discretization and integration errors that occurred in the model.

The clay soil simulation (Fig. 4.24), the diffusion dominates the system, but it is apparent that for a higher diffusion value, transport is farther than that of a lower diffusion value. Diffusion was modeled in both a horizontal and vertical gradient, therefore for a diffusion value of 0.01 m^2/y; the vertical concentration was lower than that of the other values because the solute not only travelled downward, but also horizontally. For a clay situation at 50 years and a concentration of 50%, the profile would be at a depth of approximately one meter.

The till situation shown in figure 4.25, displays a more advection dominated system. Diffusion values demonstrate that for a higher diffusion value, the contaminant plume will travel further horizontally and vertically creating a larger contaminant bulb under a burial trench. For lower diffusion values, the bulb will be smaller horizontally but have a higher concentration. At 50 years, it can be expected the center of the concentration plume will be at a depth of approximately 5 meters with 30 to 50 percent of the initial concentration depending on the diffusion value.

Sandy soils show the most significant transport of the contaminant plume in a short amount of time. Figure 4.26 represents a 10 year plot of the concentration contour. A 50 year plot could not be shown because all of the contaminant had exited the system by that point. At 10 years, the concentration reached a depth of 12 meters with the center of the plume at 40% of the initial concentration.

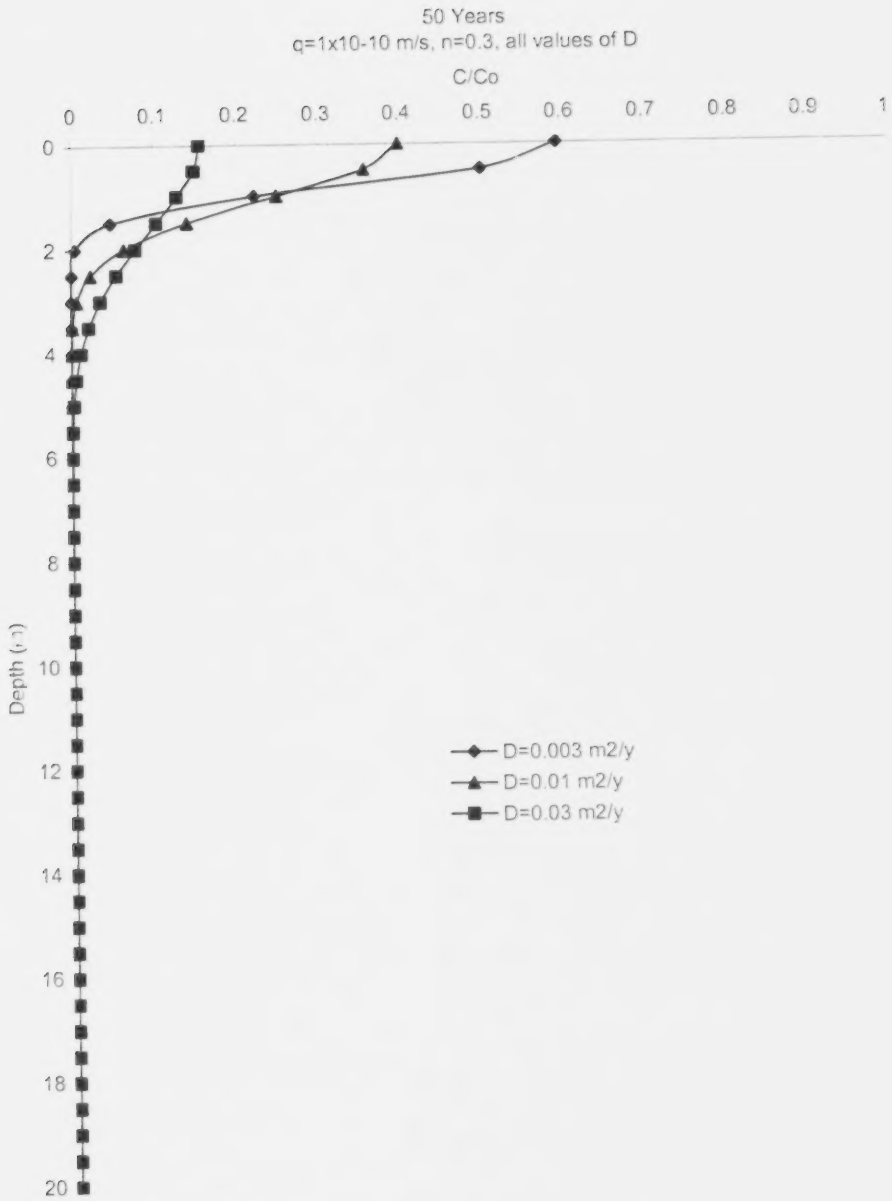

Figure 4.24 Clay soil. fifty year concentration transport profile – varied diffusion

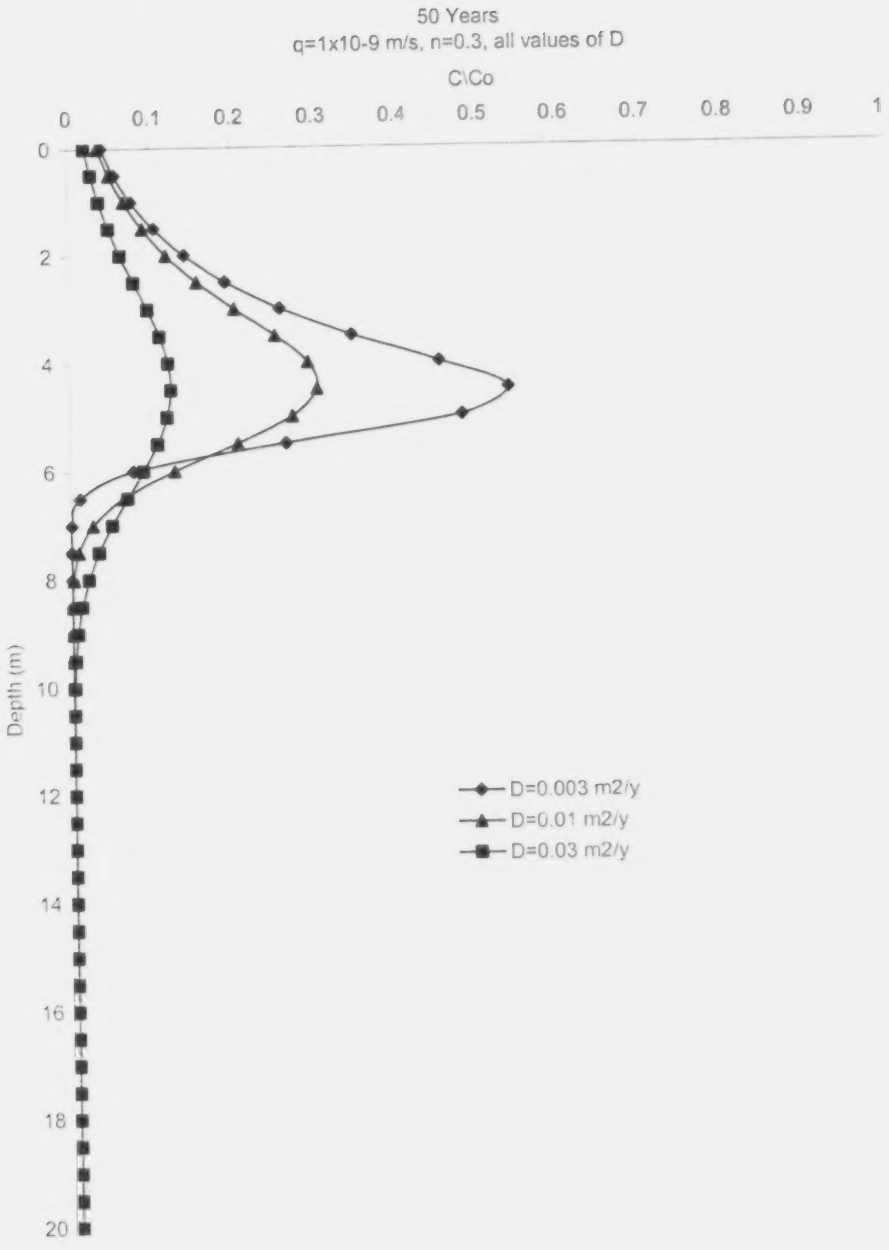

Figure 4.25 Till soil, fifty year concentration transport profile – varied diffusion

Figure 4.26 Sandy soil. ten year concentration transport profile

Porosity has changes in the transport of the contaminant plume through different soil types. For a clayey soil, porosity is shown to have little effect on the diffusion dominated system, but for situations in till and sand where advection dominates, porosity has a significant effect. Figures 4.27 – 4.29 show the results on the effect of porosity for each situation. For each situation, higher porosity corresponds to less vertical transport, while lower porosity values cause the contaminant to transport farther.

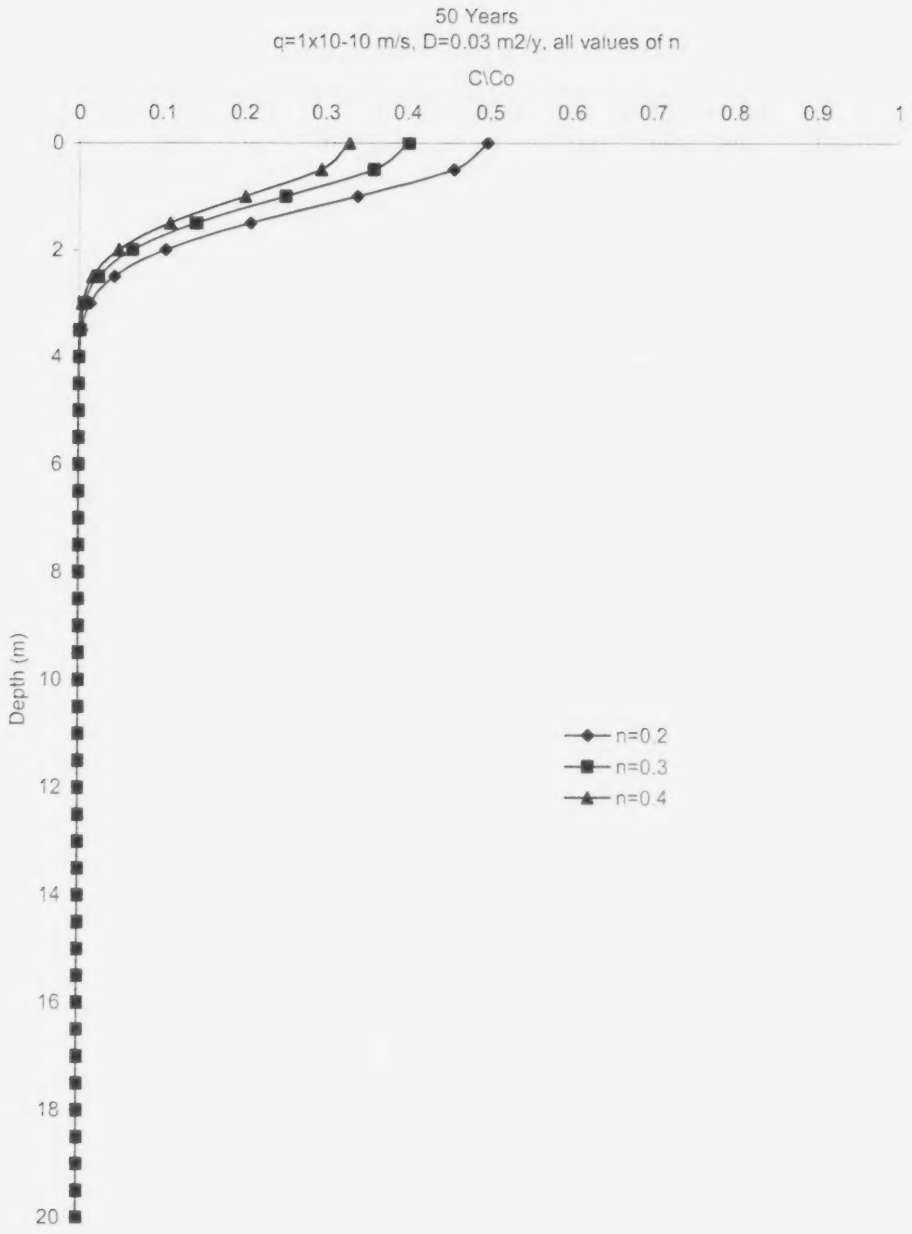

Figure 4.27 Clay soil. fifty year concentration transport profile – varied porosity

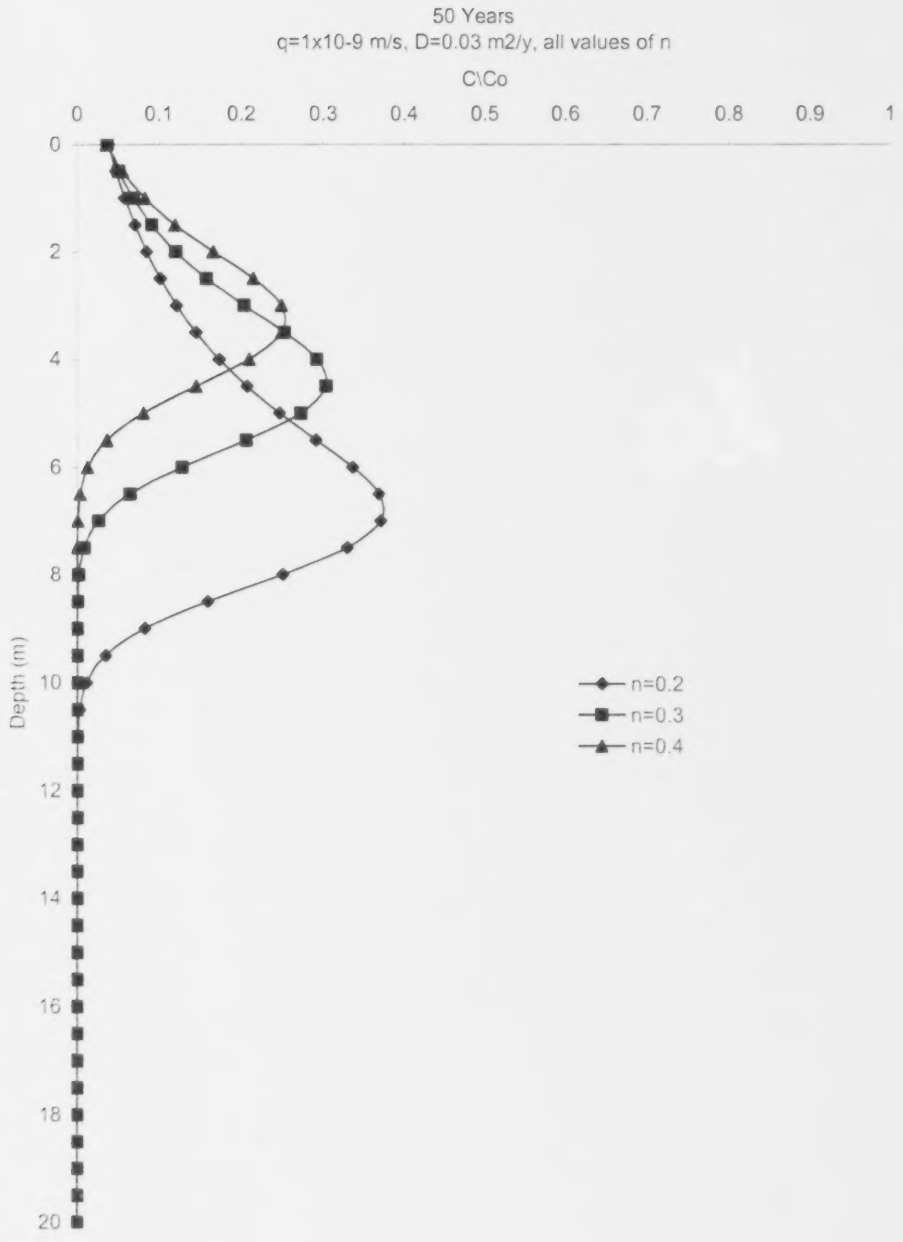

Figure 4.28 Till soil, fifty year concentration transport profile – varied porosity

Figure 4.29 Sandy soil. ten year concentration transport profile – varied porosity

Figures 4.30 – 4.32 show concentration profiles for each soil type for a diffusion value of 0.03 (0.01 for sand), and a porosity of 0.3. The clay soil has a breakthrough curve at approximately 3 meters at 10 percent of the initial concentration at 50 and 100 years. Diffusion has dominated the situation and spread the contaminant horizontally as well. For 100 years of transport in the till soil, the contaminant reached a depth of 10 meters at 10 percent of the initial concentration. In ten years, the sand reached the same depth as the till with a much higher concentration of approximately 40 percent of initial values.

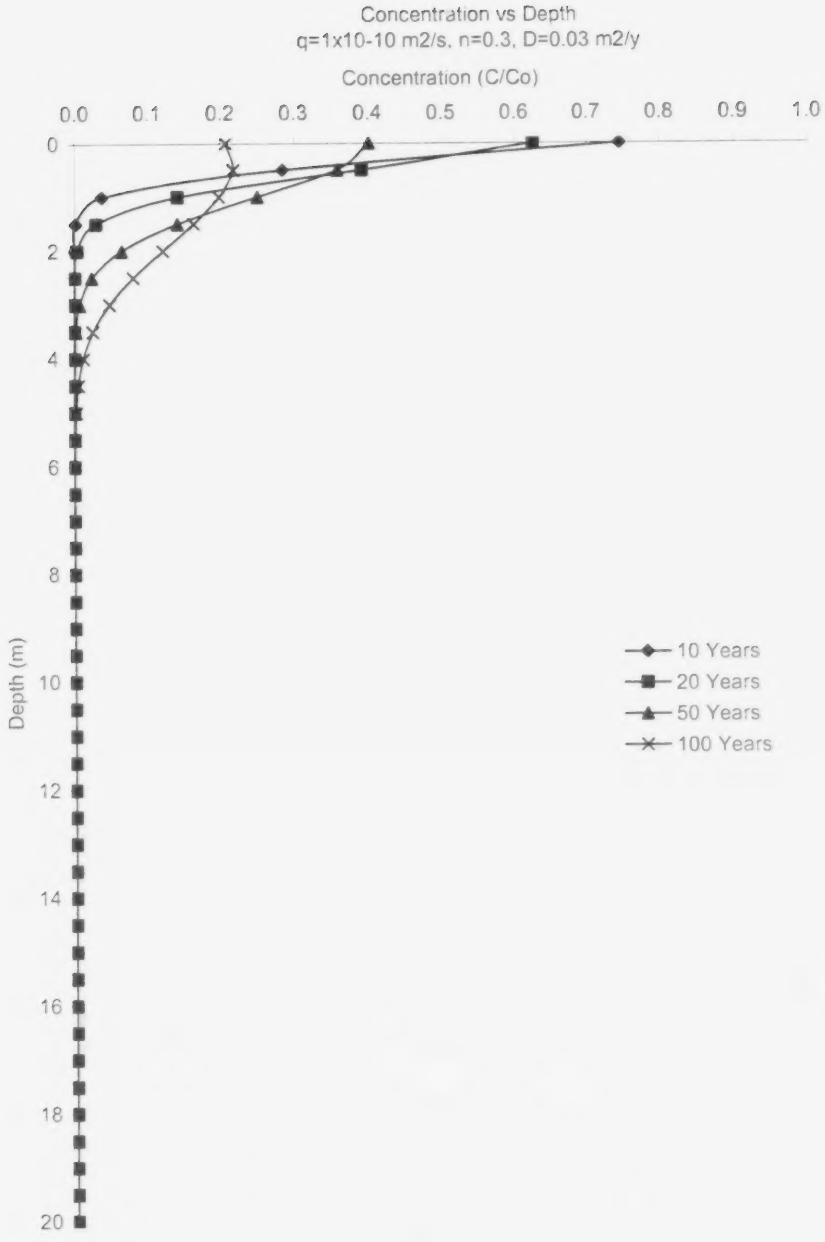

Figure 4.30 Concentration profile in clay to 100 years

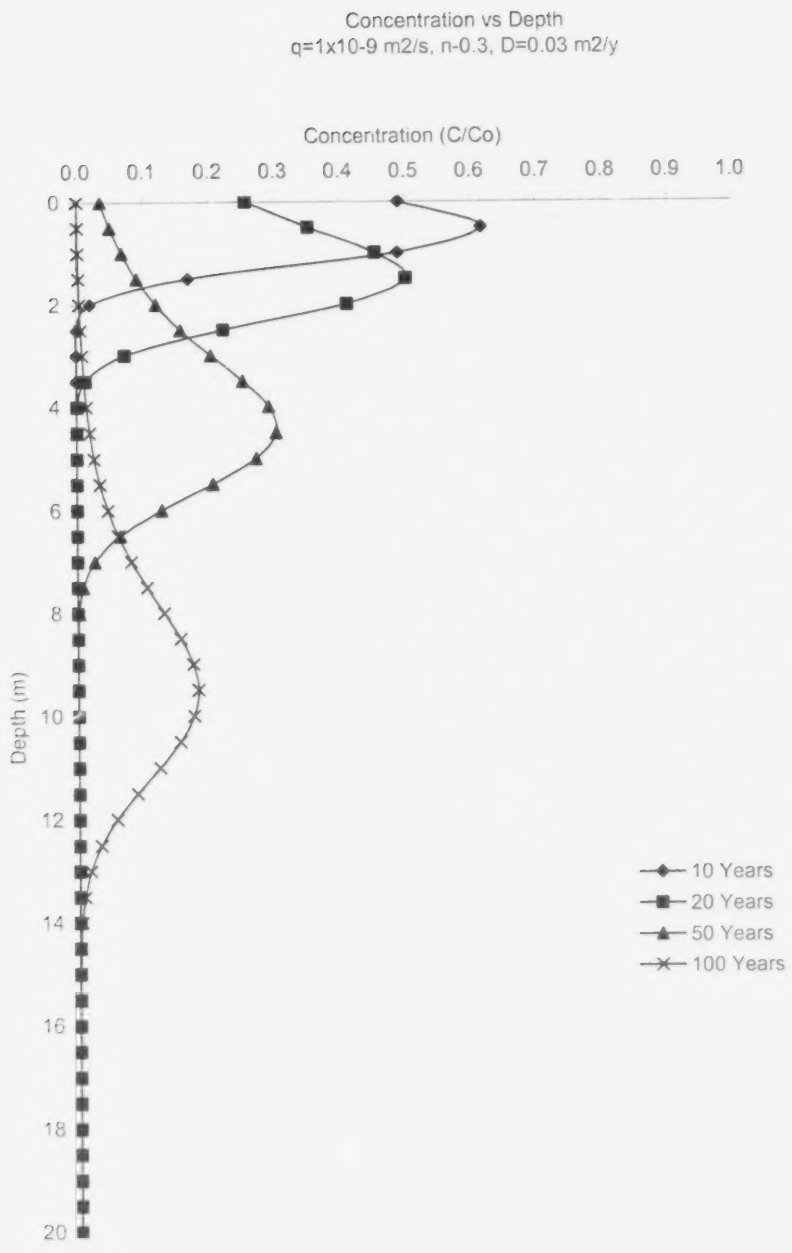

Figure 4.31 Concentration profile in till to 100 years

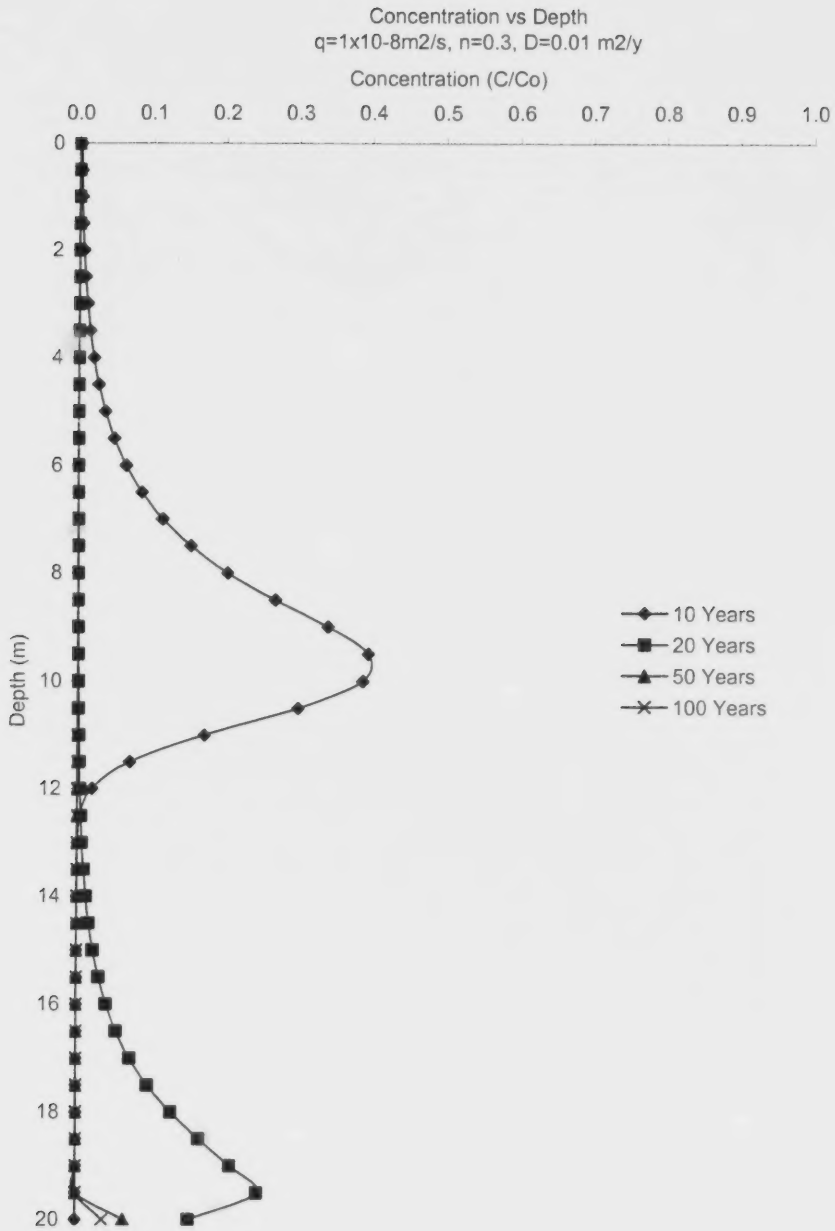

Figure 4.32 Concentration profile in sand to 100 years

The results displayed above are of key importance when looking at a potential site for livestock mortality burial. Unattenuated transport can exceed ten meters at twenty percent of initial concentration within 100 years and five meters at thirty percent concentration within 50 years.

4.3 Geo-Slope CTRAN Results

This section presents the results found from the Geo-Slope CTRAN 2004 modeling software. Concentration profiles are presented as a ratio of computed concentration versus initial concentration (C/Co). The initial concentration was assumed to be an average ammonium value of 15000 mg/l with no attenuation or adsorption included in the model. Boundary conditions were specified in section 2.4.1.3. Three models were created to simulate transport through clay, till and sand.

The concentration profiles for a clay soil (fig. 4.33) show that at 50 years, 10 percent of the initial concentration has reached a depth of three meters. The concentration profile for a till soil shown in figure 4.34 depicts 10 percent of the initial concentration will reach a depth of 10 meters in 50 years. This model has produced comparable results with the MIGRATE model results shown in the previous section. The sandy soil simulation shown in figure 4.35, demonstrates a very high transport velocity of the contaminant. At five years, the contaminant has gone just as far as it had in 50 years in a till soil. The sandy soil allows for rapid advection and dispersion causing wide horizontal and vertical spread. This is noticed later on in figure 4.3 Due to its ability to spread widely, the concentrations at depth are lower than that of the till and clay soils.

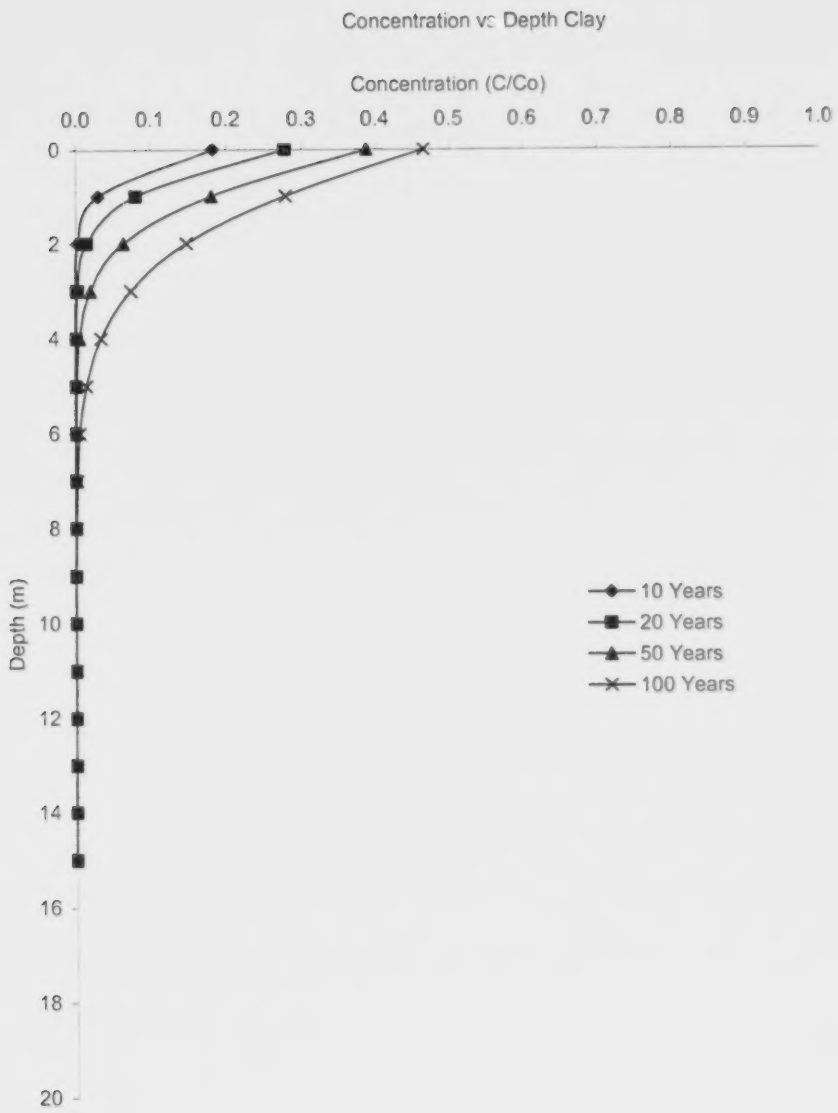

Figure 4.33 Concentration profile for clay to 100 years

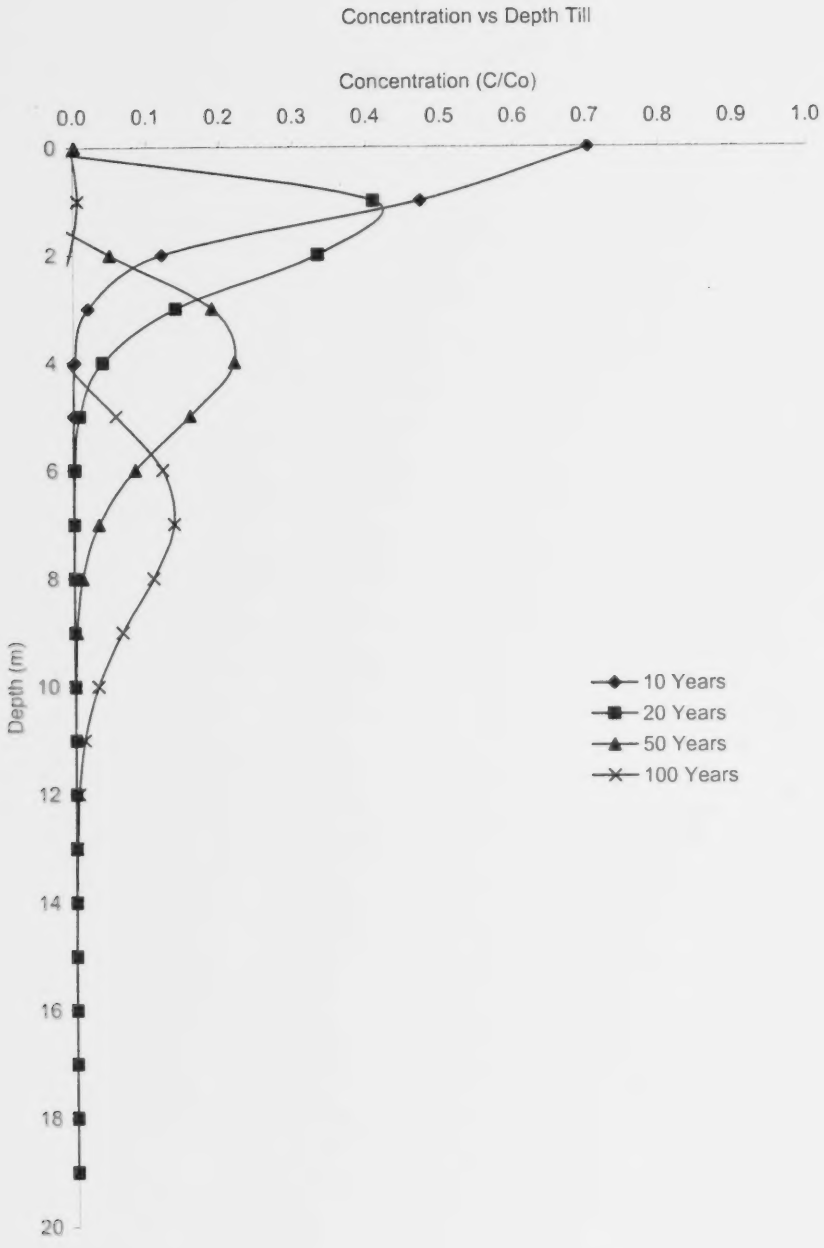

Figure 4.34 Concentration profile for till to 100 years

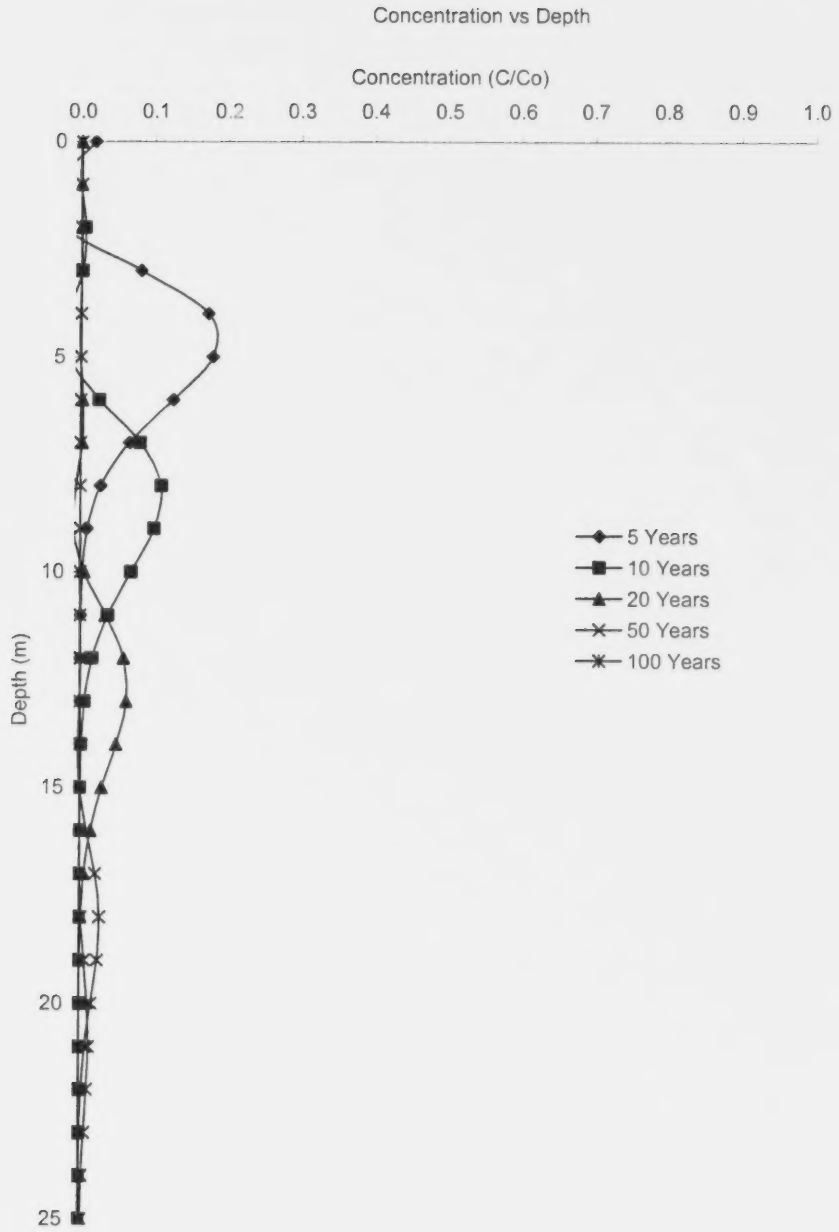

Figure 4.35 Concentration profile for sand to 100 years

The graphical interface of the Geo-Slope software package provides cross sectional

profile plots. Results for clay, till and sand are shown in figures 4.36 – 4.38. The models for clay show minimal transport of the contaminant after 10 years. Even after 100 years, the plume has only reached 4 meters at a concentration of 501 mg/l which is approximately 3 percent of the initial concentration. For till (Fig. 4.37) the contaminant plume reaches approximately 5 meters with a concentration of 1500 mg/l in a 50 year time period, which is approximately 10 percent of initial concentration. In a sandy soil, the 20 % of the initial concentration has transported approximately ten meters after only ten years. Horizontal transport is also displayed on the plots, with clay having the smallest horizontal bulb at a distance of an 8 meter diameter from the center of the pit to each side after 100 years and sand having the largest with an 8 meter diameter after only ten years.

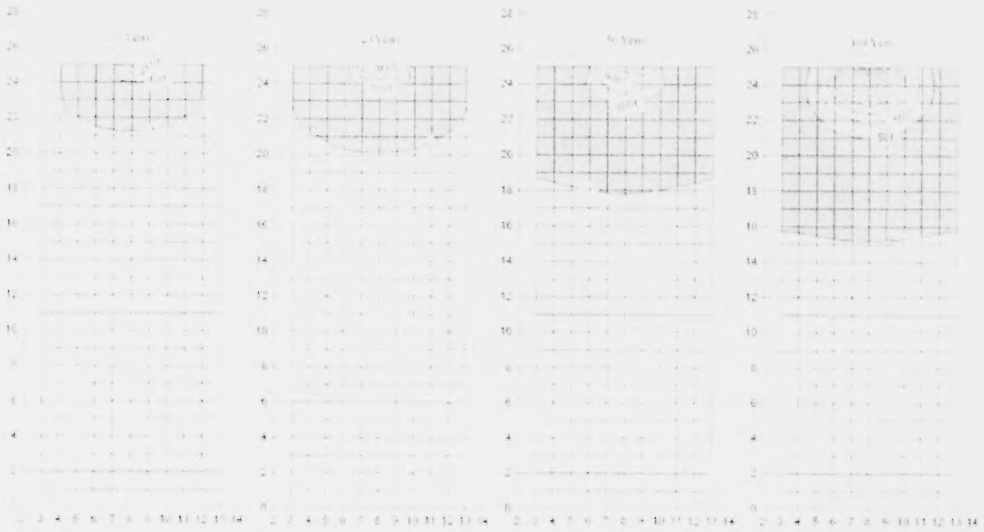

Figure 4.36 Clay soil cross sectional transport view

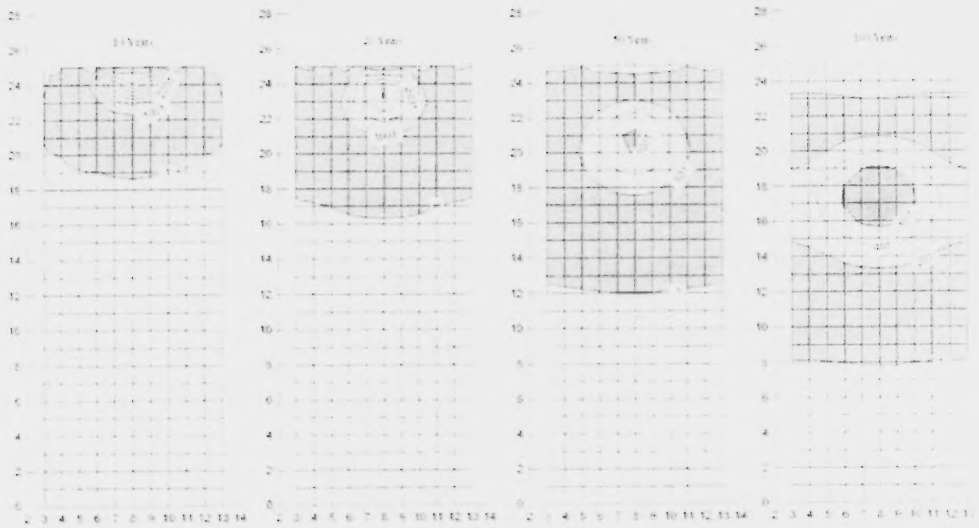

Figure 4.37 Till soil cross sectional transport view

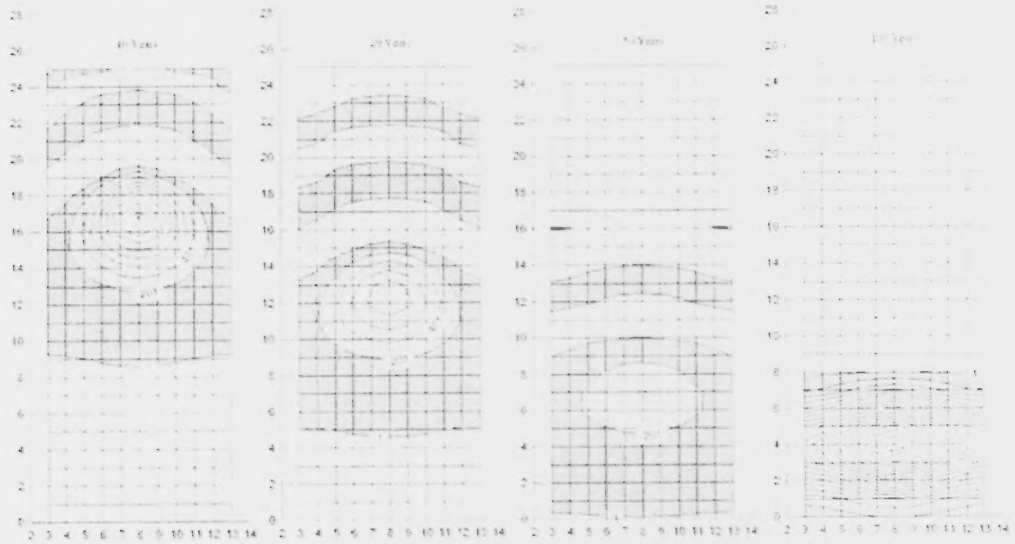

Figure 4.38 Sandy soil cross sectional transport view

4.4 Model Validation

For all models, MIGRATE has predicted a higher concentration value for each time step

but has approximately the same contaminant breakthrough curves for each soil type. For the clay soil, MIGRATE predicts a 20% concentration at 2 meters, while Geo-Slope predicts the same concentration at approximately 1.5 meters. The key importance is the contaminant breakthrough curve for clay soil is the same (Fig. 4.39). For both models the contaminant breaks through at roughly three meters with identical concentration values. Figure 4.40 represents the simulation for a till soil. MIGRATE again predicts a higher concentration value that Geo-Slope at 50 years, but it is important that both models again show the same contaminant breakthrough curve at nearly the same concentration value. The sandy soil simulation shown in figure 4.41 depicts the same scenario as the first two. MIGRATE predicts a higher concentration value than Geo-Slope, but due to MIGRATE's inability to calculate for deeper depths, it is hard to tell where the breakthrough occurs in that model.

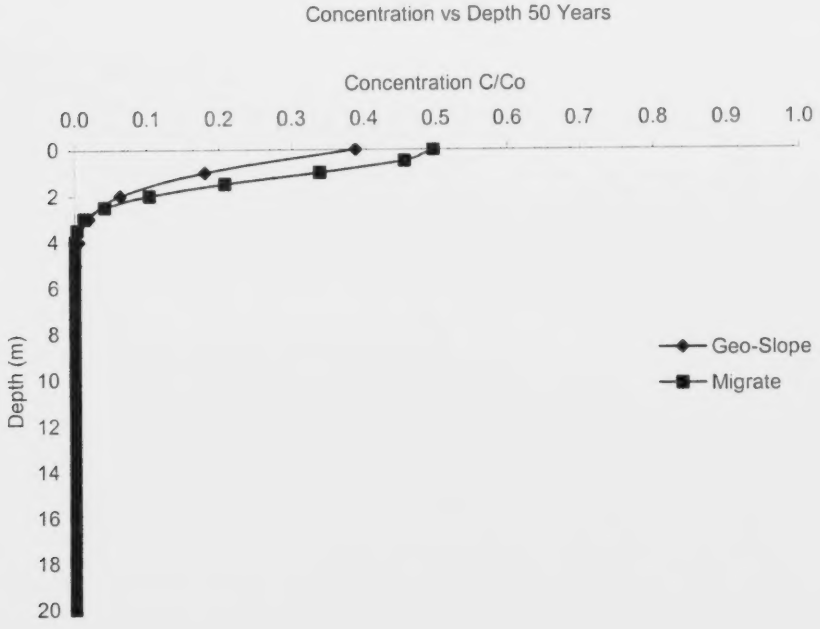

Figure 4.39 Model comparisons for clay

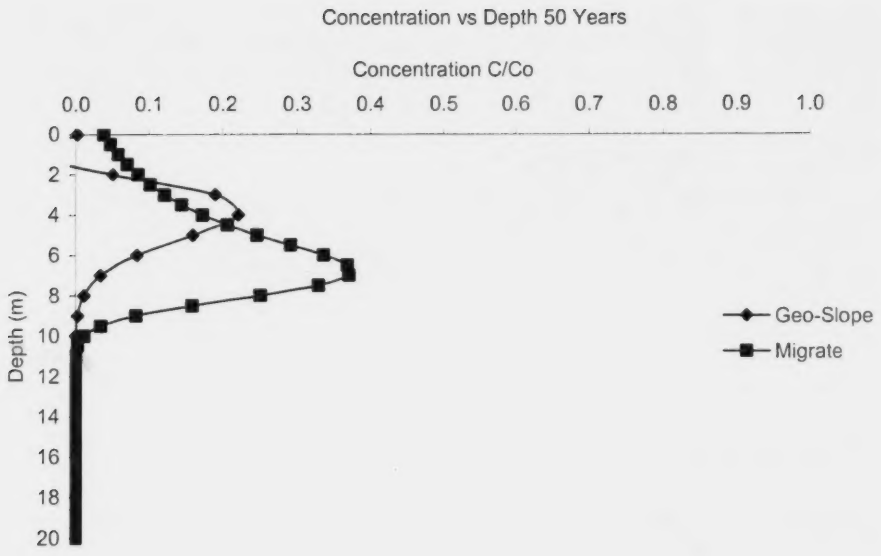
Concentration vs Depth 50 Years
Concentration C/Co
0.0 0.1 0.2 0.3 0.4 0.5 0.6 0.7 0.8 0.9 1.0
Depth (m)
0 2 4 6 8 10 12 14 16 18 20
Geo-Slope
Migrate

Figure 4.40 Model comparisons for till

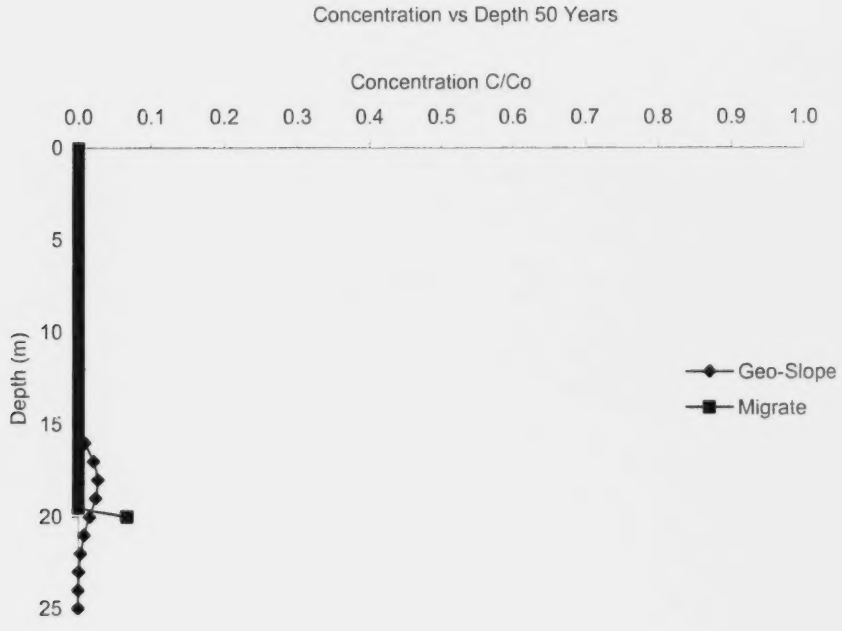

Figure 4.41 Model comparisons for sand

4.5 Model Significance and Limitations

Each model produced from MIGRATE and Geo-Slope demonstrates a worst case scenario for three disposal soil types. Convergence issues occurred in the integration steps for both models when calculating for a sand situation. The MIGRATE model was only capable of performing the integration without errors for a diffusion value of 0.01 m^2/y and Geo-Slope used a value of 0.03 m^2/y for all soil situations, with slight convergence issues for the sand situation as well.

By using software based on two-dimensional equation to predict contaminant transport only gives an estimate of the situation at hand. Real life conditions such as non-homogeneous soils and horizontal velocity gradients were not modelled in this thesis due to their complexity. These models also do not consider infiltration of rainwater, draught conditions, or other types of water flux. The Geo-Slope package is capable of simulating these properties if they are known for the modelled time period.

Simulating a worst case scenario, by overestimating leachate amounts and locating the burial pit bottom in the water table, gives a good estimate of what transport times and concentrations to expect. Also, by modeling a conservative contaminant, attenuation and adsorption are not considered in the model and therefore in a real life setting, the transport concentrations and distances will be smaller than expected.

5 CONCLUSIONS

The objectives of this thesis were to determine the chemistry of livestock mortality leachate for three species: bovine, swine and poultry. By characterizing the leachate from the mortalities, models were created to simulate a worst-case scenario of leachate transport for three soil conditions a clay, till and sand. By creating contaminant transport models, recommendations for burial could be determined.

Livestock mortality leachate contains on average a concentration of 9000 mg/l of ammonium, 31000 mg/l alkalinity, 2500 mg/l chloride, 4500 mg/l sulphate, 2000 mg/l potassium, 1900 mg/l sodium, 1700 mg/l phosphorus along with small amounts of iron, calcium and magnesium. The pH of the leachate was near neutral. In comparison to earthen manure storages and landfills, the strength of the leachate was much higher.

Preliminary simulations were created with two software packages, MIGRATEv9 and Geo-Slope CTRAN to simulate transport of the leachate for three different soil settings. Transport through a clayey soil was dominated by diffusion causing the leachate to only transport a distance of approximately three meters in 50 years. The till soil situation produced a transport depth of six meters with an approximate concentration of thirty percent of initial values. For a sandy soil, transport reached a depth of 10 meters in 10 years with approximately forty percent of initial values. The MIGRATE and Geo-Slope

models produced similar results with the MIGRATE model predicting slightly higher concentration values than that of Geo-Slope.

The results of this thesis should aid in determining burial regulations by providing a basic understanding of the composition of leachate arising from a mortality disposal site. The worst case scenario models can provide a basic guideline for similar soil types.

6 RECOMMENDATIONS FOR FURTHER RESEARCH

Due to the strength of livestock mortality leachate, the option of burial locations should be carefully determined. Further research would be helpful to determine the fate of the bacteria or viruses causing the diseases in the livestock and whether or not it is transported through the underlying soil. Research on existing burial sites should be conducted to study how far the leachate has travelled since burial and how much has been attenuated by the soil particles.

7 LIST OF REFERENCES

Appelo, C. A. J. and D. Postma (1996). Geochemistry, Groundwater and Pollution. Rotterdam, A.A. Balkema Publishers.

Canadian Food Inspection Agency. 2006. *Burial Procedural Plan Draft 1.* Disease Control – Programs Branch, Animal Production and Health Division.

Childs, P.E. 2001. Phosphorus: from urine to fire. *Chemistry in Action!* 64:32.

Engel, B.A., Lim, K.J., Choi, J.Y., and L. Theller. 2004. Chapter 14: Evaluating Environmenal Impacts. *Carcass Disposal: A Comprehensive Review.* National Agricultural Biosecurity Center Consortium USDA. Kansas State University.

Environment Agency. 2001. The environmental impact of the foot and mouth disease outbreak: an interim assessment. *Environment Agency*, ISBN 1-85-7057856.

Environment Canada. 2004. Groundwater [Online] http://www.ec.gc.ca/water/en/nature/grdwtr/e_gdwtr.htm (posted 06/08/2004; verified 10/25/2005).

Fetter, C. W. (1999). Contaminant Hydrogeology. Upper Saddle River, NJ, Prentice-Hall.

Fonstad, T.A. (2004). Transport and Fate of Nitrogen from Earthen Manure Storage Effluent Seepage. Ph.D. Thesis, University of Saskatchewan. Saskatoon, SK.

Fonstad, T.A., Meier, D.E., Ingram, L.J., and J Leonard. 2003. Evaluation and demonstration of composting as an option for dead animal management in Saskatchewan. Canadian Biosystems Engineering Volume 45.

Freeze, R.A., and J.A. Cherry. 1979. Groundwater. Englewood Cliffs, NJ. Prentice Hall.

Gelhar, L.C. Welty, et al. 1992. A Critical Review of Data on Field-Scale Dispersion in Aquifers. *Water Resources Research.* 28(7): 1995-1974.

Glanville, T. 2000. Impact of Livestock Burial on Shallow Groundwater Quality. Proceedings of the American Society of Agricultural Engineers, Mid Central Meeting, St. Joseph, MI, ASAE.

Hendry, M.J. 1982. Hydraulic Conductivity of a Glacial Till in Alberta. *Groundwater.* 20(2): 162-169.

Jansen, Andy. August 11, 2006. Personal Communication. Saskatchewan Agriculture and Food.

Kalbasi, A., Mukhtar, S., Hawkins, S.E., and B.W. Auvermann. 2005. Carcass Composting for Management of Farm Mortalities: A Review. *Compost Science & Utilization.* 13(3):180-193.

Kastner, J. and Randall Phebus. 2004. Chapter 2: Incineration. *Carcass Disposal: A Comprehensive Review.* National Agricultural Biosecurity Center Consortium USDA. Kansas State University.

Keller, K., G. van der Kamp, et al. 1986. Fracture Permeability and Groundwater Flow n Clayey Till Near Saskatoon, Saskatchewan. Canadian Geotechnical Journal. 23: 229-240.

Krahn, John. 2004. Transport Modeling with CTRAN/W. Geo-Slope International. Calgary, Alberta.

Krahn, John. 2004 Seepage Modeling with SEEP/W. Geo-Slope International. Calgary, Alberta.

MacArthur, A.J., Milne, J.C., and P.J. Young. 2002. Leachate Characteristics Arising from the Foot and Mouth Mass Burial Site in Scotland. <http://www.leachate.co.uk/Foot_and_Mouth_Disease_Leachate_Paper.pdf> (15 June 2005)

MacArthur, A.J. and J.C. Milne. 2002. Leachate Characteristics and Management Requirements Arising from the Foot & Mouth Operations in Scotland. Proceedings Waste 2002. Integrated Waste Management and Pollution Control: Research, Policy and Practice: 305-314.

Mitchell, H.H., Hamilton, T.S., Steggerda, F.R., and H.W. Bean. 1945. The Chemical Composition of the Adult Human Body and its Bearing on the Biochemistry of Growth. *The Journal of Biological Chemistry*. 158:3, 625-637.

Moore, J.W., Statinski, C.L. and P.C. Jurs. 2002. Chemistry: The Molecular Science 1st Edition. Thomson Learning Inc.

Palmer, C. (1996). Contaminant Hydrogeology. Boca Raton, FL, CRC Press Inc.

Parkhurst, D.L. & C.J. Appelo. 1999. Users Guide to Phreeqc (Version 2) – A Computer Program for Speciation, Batch Reaction, One-Dimensional Transport, and Inverse Geochemical Calculations. *Water Resources Investigations Report 99-4259*. U.S. Geological Survey.

Rail, C. (1989). Groundwater Contamination. Lancaster, PA, Technomic Publishing Company, Inc.

Ritter, W. F. and A. E. M. Chirnside. 1995. Impact of Dead Bird Disposal Pits on Groundwater Quality on the Delmarva Peninsula. *Bioresource Technology* (53): 105-111.

Rowe, R.K. 1990. Background Documentation for Programs POLLUTE & MIGRATE. Geotechnical Research Center. Report # GEOP 90-2.

Rowe, R.K., Booker, J.R. and M.J. Fraser. 1995. MIGRATEv9 Users Guide. GAEA Environmental Engineering Ltd. Ontario, Canada.

Saskatchewan Agriculture and Food (SAF). Farm Facts: Managing Livestock Mortalities.2005. <http://www.agr.gov.sk.ca/docs/production/managinglivestockmortalities.asp>

(15 Oct, 2006)

Scudamore, J.M., Trevelyan, G.M., Tas, M.V., Varley, E.M. and G.A.W. Hickman. 2002. Carcass Disposal: lessons from Great Britain following the foot and mouth disease outbreaks of 2001. Rev. sci. tech. Off. Int. Epiz. 2002, 21(3): 775-787.

Sherman, H.C. and E.J. Quinn. 1926. The Phosphorus Content of the Body in Relation to Age, Growth and Food. *The Journal of Biological Chemistry*. 67:3. 667-677.

SSC. (2003a). European Commission Scientific Steering Committee Final Opinion and Report on a Treatment of Animal Waste by Means of High Temperature and High Pressure Alkaline Hydrolysis. European Commission: Health and Consumer Protection Directorate-General.

Statistics Canada. Statistics by Subject: Summary tables in Canadian Statistics on Livestock and Aquaculture. <http://www40.statcan.ca/l01/cst01/prim50i.htm> (24 Jan. 2007)

Thornton, S.F., Lerner, D.N., and Tellam, J.H. 2001. Attenuation of landfill leachate by clay liner materials in laboratory columns: 2. Behaviour of inorganic contaminants. Waste Management and Research, 19: 70-88.

Van Genuchten, M.T. 1980. A Comparison of Numerical Solutions of the One-Dimensional Unsaturated-Saturated Flow and Mass Transport Equations. *Finite Elements in Water Resources Volume 1*. Mississippi University, Oxford.

Vass, A.A. 2001. Beyond the Grave – Understanding Human Decomposition. *Microbiology Today*. Vol. 28/Nov.

APPENDIX A – SRC Analytical Laboratory - Leachate Analysis Methods

LEACHATE ANALYSIS

Ammonia

To analyze for ammonia, the water automated colorimetric method was used. Ammonia reacts with hypochlorite and phenol in the presence of a catalyst to form idophenol, which gives the sample a blue color. The Cobras Fara II automated centrifugal analyzer measured the absorbance of light in the solution at 630 nm which is proportional to the concentration. The detection limit is 0.01 mg/l. Samples with concentrations greater than 2.0 mg/l must be diluted.

Nitrate and Nitrite

Colorimetric determination with a continuous flow analyzer was used to determine Nitrate and Nitrite in the leachate. Nitrate is reduced to Nitrite with hydrazine sulphate and the nitrite is determined by diazotizing with sulphanilamide and coupling with N-(1-naphthyl)-ethylenediamine dihydrochloride to form a highly coloured azo dye which is measured colorimetrically at 529 nm. Nitrite was analyzed without the reducing agents using the colorimetric technique. The detection limit is 0.01 mg/l as N. Samples with concentrations above 2 mg/l must be diluted.

Chloride

Chloride was analyzed using the mercuric thiocyanate colorimetric method on the Cobras Fara II centrifugal analyzer. The thiocyanate ion (SCN-) is displaced from mercuric thiocyanate by chloride. The liberated thiocyanate ion reacts with ferric ion to form the coloured complex ferric thiocyanate. The concentration is proportional to the original chloride concentration. The concentration is measured by measuring the absorbance at 480 nm. The detection limit was 1 mg/l. Samples with concentrations

greater than 500 mg/l must be diluted.

Organic Carbon

Organic carbon in the leachate was analyzed using the Dohrmann Phoenix 8000 carbon analyzer. Aqueous samples were treated with phosphoric acid to release inorganic carbon as CO_2. The remaining organic carbon in the sample was reacted with sodium persulphate and UV light to release the organic carbon as CO2. The CO2 was measured with a non-dispersive infrared detector. The detection limit was 0.2 mg/l for both Total Organic Carbon and Dissolved Organic Carbon. Samples with concentrations greater than 20 mg/l must be diluted.

Calcium, Magnesium, Potassium, Sodium and Sulphate

Several elements in the leachate were determined using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) using the TJA-IRIS Advantage Analyzer. The plasma was a stream of argon gas ionized by an applied radio frequency field. Sample aerosols were injected into the plasma which subjected the atoms to temperatures of 6000 to 10,000 K. The high temperature ionized a high percentage of atoms in the aerosol and reduced chemical interferences. The emission spectra of the ionized atoms were optically measured in a computer controlled spectrometer. The light emitted from the plasma was focused onto the entrance slit of and Echelle Optical System. A prism and diffraction grating disperse the light which was focused on a Charge Injection Device (CID) detector. The detector monitored all configured wavelengths in a computer controlled environment. Upper detection limits for these elements are as follows: Calcium 50 mg/l, Magnesium 100 mg/l, Potassium 20 mg/l, Sodium 15 mg/l and Sulphate 300 mg/l.

Metals

Several elements in the leachate were determined using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) using the TJA-IRIS Advantage Analyzer. The plasma was a stream of argon gas ionized by an applied radio frequency field. Sample aerosols were injected into the plasma which subjected the atoms to temperatures of 6000 to 10,000 K. The high temperature ionized a high percentage of atoms in the aerosol and reduced chemical interferences. The emission spectra of the

ionized atoms were optically measured in a computer controlled spectrometer. The detection limits and upper limits for this procedure are shown in Table A1.

Table A1: Detection Limits of Various Metals

Metal	Upper Limit (mg/L)	Detection Limit (mg/L)
Al	70	0.5
Ba	30	0.1
Be	10	0.1
B	100	0.2
Cd	20	0.1
Cr	100	0.1
Co	100	0.1
Cu	60	0.1
Fe	50	0.1
Pb	35	0.2
Mn	30	10
Mo	50	0.1
Ni	60	0.1
P	50	1
Si	100	1
Ag	50	0.1
Sr	10	0.1
Ti	30	0.1
V	100	0.1
Zn	30	0.5
Zr	70	0.1

Table A2: Summary of detection limits (units of mg/L)

Inorganic Chemistry	Detection Limit	Upper Limit
Bicarbonate	1	
Carbonate	1	
Chloride	1	500
Hydroxide	1	
pH	0.07	
Alkalinity	1	
Ammonia-N	0.01	2
Nitrate-Nitrite as N	0.01	2
TKN	0.05	2
Inorganic Carbon	0	
Organic Carbon	0.2	20
Sulfate	10	300
S	5	
K	0.1	20
Ca	1	50
Al	70	0.5
Ba	30	0.1
Be	10	0.1
B	100	0.2
Cd	20	0.1
Cr	100	0.1
Co	100	0.1
Cu	60	0.1
Fe	50	0.1
Pb	35	0.2
Mn	30	10
Mo	50	0.1
Ni	60	0.1
P	50	1
Si	100	1
Ag	50	0.1
Sr	10	0.1
Ti	30	0.1
V	100	0.1
Zn	30	0.5
Zr	70	0.1

APPENDIX B – Project Pictures

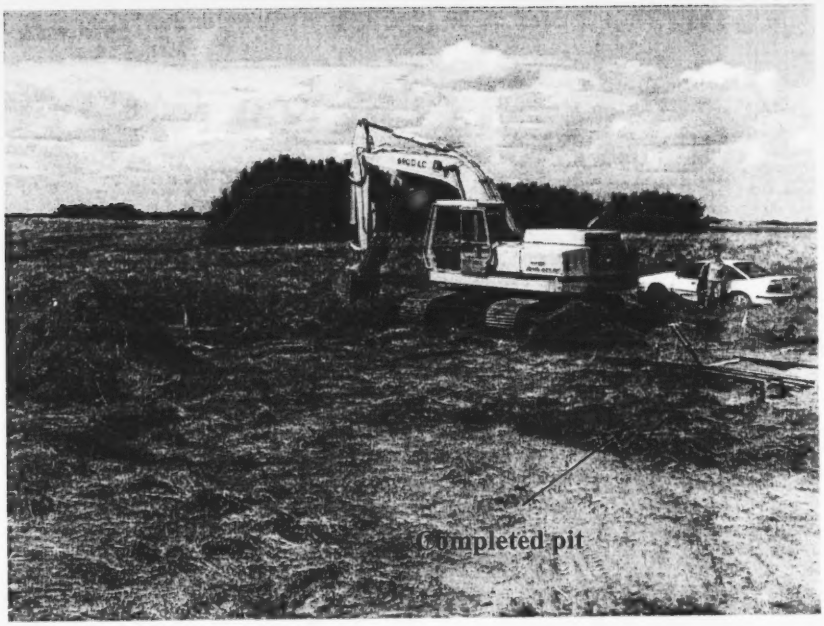

Figure B.1 – Excavation of pits

Figure B.2 – HDPE Liner and Leachate Collection System

Figure B.3 – Leachate collection system with recirculation/sampling tubing installed

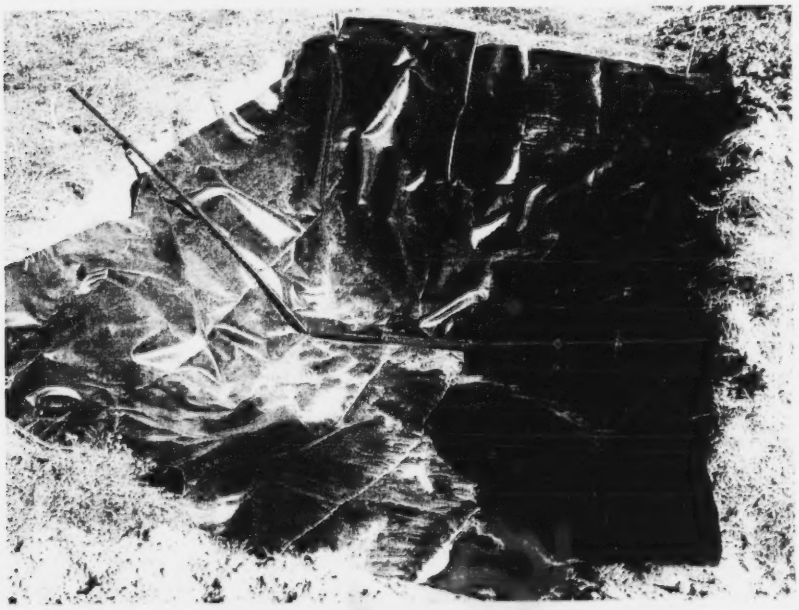

Figure B.4 – Liner in pits with only mortalities (HDPE liner used to cover mortalities prior to soil cover)

Figure B.5 – Swine mortalities in pits with soil and no HDPE cover

Figure B.6 – Soil cover installation in swine mortalities pit

Figure B.7 – Bovine mortalities in pits with soil and no HDPE cover

Figure B.8 – Soil cover installation in bovine mortalities pit

Figure B.9 – HDPE cover installation with soil cover and venting tubes

Figure B.10 – Completed pits with monitoring tube access and venting

Figure B.11 – Sampling from leachate collection system. (Leachate was first circulated to ensure well mixed sample and then the tubing redirected to a 1L sample bottle)